

**ECOHYDROLOGICAL IMPACTS OF CLIMATE
CHANGE ON A RIPARIAN CHALK VALLEY
WETLAND**

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Declaration

I, Andrew House, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

A handwritten signature in black ink, appearing to read 'A. House', is written over a light gray dotted rectangular background.

15th April 2016

Abstract

This thesis assesses the impacts of climate change on the CEH River Lambourn Observatory, Boxford, UK. This comprises a 10 ha chalk valley, riparian wetland and 600 m of the River Lambourn, designated for its conservation value and scientific interest. A field campaign targeted knowledge gaps in previous research to enable development of a conceptual model of hydrological functioning. The physically based, distributed model MIKE SHE was chosen to simulate hydrology due to flexibility in process representation and proven applicability to wetland hydrology. Model results were consistent with field observations and confirmed the conceptual model. Findings showed that groundwater/surface-water interaction dominates hydrological processes. Channel head boundaries broadly control water levels across the wetland. Areas of groundwater upwelling control discrete head elevations and contain high concentrations of nitrate. These support confined growth of *Carex paniculata* surrounded by poor fen communities in reducing higher-phosphate waters. In-channel macrophyte growth and its management through cutting acutely affect water levels. Impacts of climate change were assessed by driving the MIKE SHE model with projected changes in hydrometeorological inputs for the 2080s, derived from UKCP09. Areas of groundwater upwelling caused amplified response of water levels at distinct locations. Simulated water levels were linked to requirements of the MG8 plant community and Desmoulin's whorl snail (*Vertigo moulinsiana*). Impacts on each differed spatially, in line with hydrological impacts. The PHABSIM habitat modelling methodology was modified to assess river habitat response for brown trout (*Salmo trutta*), using outputs from the 1D hydraulic component of MIKE SHE, MIKE 11. Reductions in habitat availability were pronounced through periods of low flows, more so for adult than juvenile trout. Different hydrological requirements for species in distinct areas of the site support separate management strategies. Multiple objective management may be achieved through adaptive modification of the current management regime.

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Chapter 1

The hydrology and hydroecology of groundwater dependent wetlands

1.1 Introduction

This thesis investigates the impacts of climate change on the hydrology and hydroecology of the CEH River Lambourn Observatory, a groundwater dependent floodplain wetland in the chalk lowlands of southeast UK. Such wetlands are widely established along river valley bottoms across the freely draining chalk, despite the permeability of the bedrock. They have a rich history of cultivation as water meadows (Everard, 2005), and, as a consequence of their unique setting, provide habitats for diverse species important to conservation (Fojt, 1994). Hence, many are protected under national and international legislation. Yet, water fluxes in these areas, and how they link to ecological processes, remain poorly understood.

In recent times, wetlands have come to be widely recognised as providing valuable environmental, cultural and economic functions and services (Acreman et al., 2011b). Occupying ambiguous transitional zones between wet and dry areas, they share characteristics of both aquatic and terrestrial environments, yet cannot be consigned to either category (Ramsar, 2011). The European Habitats Directive (EEC, 1992) lists groundwater dependent wetland ecosystems as priority habitats that are particularly sensitive to environmental change. The need for sustainable wetland management is intensifying in the face of climate change as well as growing, and often competing, demands for water (Maltby and Acreman, 2011).

The establishment and maintenance of wetlands depends primarily on the hydrological regime (Mitsch and Gosselink, 2007), as it is a key control on vegetation (Baldwin et al., 2001; Wheeler et al., 2009), fauna (Ausden et al., 2001; McMenamin et al., 2008) and biogeochemical cycling (Lischeid et al., 2007; McClain et al., 2003). Current and historical wetland management practices revolve around the maintenance of water levels required for the conservation of desired species or communities, flood mitigation, and arable or pastoral productivity (Morris et al., 2008). An ability to predict the impacts of modifications to wetlands' hydrological regimes is, therefore, highly desirable. Models that can accurately represent wetland hydrological processes have enormous potential in the assessment of possible degradation to the ecological character of wetlands and

for the development of wetland management schemes that could mitigate these impacts. (Acreman and Jose, 2000).

In this introductory chapter, prevalent definitions of wetlands, their extent, and value are reviewed, before progressing to the interactions between wetlands, hydrology, and ecology. A specific focus is the groundwater dependent wetlands in the chalk lowlands of the United Kingdom. The potential role of hydrological models in assessing the impacts of environmental changes, in particular climate change, on the hydrology and ecology of such wetlands is also discussed. The aim and objectives of this study are presented and the structure of the thesis outlined.

1.2 Wetlands, their extent and value

1.2.1 Definition and classification

The term ‘wetland’ was itself not commonplace in the scientific sphere until the second half of the twentieth century (Lewis, 1995). Previous terms such as bog, fen and mire were used colloquially, and these terms are now used to describe specific types of wetland. Indeed, the general term most likely developed from a need to characterise and value land, and to manage and regulate wetland ecosystems (Cowardin et al., 1979). Internationally, due to their diversity, there is no single indisputable definition, with great variety according to purpose. The International Convention on Wetlands of International Importance (Ramsar, 2011), takes a broad view, defining wetlands under the text of the Convention (Article 1.1) as:

“areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres.”

Additionally, to protect coherent sites, Article 2.1 of the Convention states that wetlands to be included in the Ramsar List of internationally important wetlands:

“may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands.”

This provides one of the most inclusive and flexible definitions and demonstrates the Ramsar Convention’s purpose to conserve all wetlands through local, regional and national actions and international cooperation, so to help achieve worldwide sustainable development. More specific definitions have been developed, significantly by organisations in the US where debates over wetland definition and classification have a

long history (Lewis, 2001) (Table 1-1). In the UK there is also no commonly accepted definition.

Traditionally, wetlands have been known by type through various nomenclatures, such as bogs, fens, marshes, swamps, mires, sloughs and moors. Attempts have been made to standardise these terms, in particular according to vegetation composition (Rodwell, 1994-2000). The usefulness of this approach, however, is restricted by regional

Table 1-1. Prevalent US definitions for wetlands

Organisation and source	Definition
US Fish and Wildlife Service Shaw and Fredine (1956)	"The term "wetlands," as used in this report and in the wildlife field generally, refers to lowlands covered with shallow and sometimes temporary or intermittent waters. They are referred to by such names as marshes, swamps, bogs, wet meadows, potholes, sloughs, and river-overflow lands. Shallow lakes and ponds, usually with emergent vegetation as a conspicuous feature, are included in the definition, but the permanent waters of streams, reservoirs, and deep lakes are not included. Neither are water areas that are so temporary as to have little or no effect on the development of moist-soil vegetation."
US Fish and Wildlife Service Cowardin et al. (1979)	"Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water... wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year."
US Army Corps of Engineers, and Environmental Protection Agency Environmental Laboratory (1987)	"Those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas."

etymologies and a lack of definition between attributes across wetland types (Finlayson and Van der Valk, 1995). Though, fens are commonly defined as receiving water and nutrients from surface and/or groundwater along with precipitation, while bogs are normally defined as receiving water exclusively from precipitation (McBride et al., 2011). More robust systems of classification promote the importance of hydrology and its relation to other geological, morphological or ecological settings (e.g. Brinson, 1993a; Brinson, 1993b). Despite a myriad of conflicting terms, three indicative features of wetlands may be recognised within most wetland definitions and also feature within the myriad of classification schemes which have been developed (Maltby and Acreman, 2011; Mitsch and Gosselink, 2007): i) the presence and flux of water either above or at ground level or within the rootzone; ii) unique soil characteristics differing from adjacent dryland areas; and iii) vegetation specifically adapted to permanent or fluctuating wet conditions.

Guidance of national and international policy requires standardised and easily understood terms (Finlayson and Van der Valk, 1995; Scott and Jones, 1995). Internationally, a global classification should provide the legal framework for wetland conservation and facilitate dissemination of information. The Ramsar Convention, contains a prescribed classification approved by member states (Ramsar, 2011). Five major wetland types are generally recognized, comprising marine, estuarine, lacustrine, riverine, and palustrine wetlands. Human-made wetlands are also recognised. The classification is very broad in nature, encompassing mangroves to karstic lakes to wastewater treatment ponds, and is intended to promote global wetland conservation.

Systems of national classification are long established in the USA, yet less so elsewhere. Older classification systems in the USA typify wetlands by comparing vegetation (Stewart and Kantrud, 1971) or soil material (Cowardin et al., 1979) to water permanence. Such methods do not account for hydrological mechanisms, morphology, topography or location (Brooks et al., 2011), omissions addressed by Novitzki's (1978) classification which employs water source and morphology. To better recognise biogeochemical functions, Brinson (1993a) applied environmental gradients to the classification of wetlands. These included descriptors of size and position, water source variations, and nutrient and sediment flow pathways. Separately, Brinson (1993b) combined indicators of geomorphic setting, water source and hydrodynamics. Classification of wetlands according to hydrogeomorphology was seen to assist assessment of their physical, chemical and biological functions. Elsewhere, in Australia, Semeniuk and Semeniuk (1995) used landform and hydrology to identify thirteen primary types of wetlands. Landform setting was combined with the degree of wetness, namely permanent,

seasonal, intermittent inundation or seasonal waterlogging, to produce a classification for use regardless of climate or vegetation.

The UK contains a variety of wetland types, which reflect a range of hydrological, geological and topographical conditions, along with the climatological gradient from a wetter west to drier east (Hughes and Heathwaite, 1995). An early attempt at a national classification system was that of Goode (1972) incorporated into the Nature Conservation Review (Ratcliffe, 1977) and developed slightly by Wheeler (1984). Hydrological and morphological characteristics were used to define major classes of wetland:

1. Flood-plain mire – developed on alluvium and occupying a wide range of nutrient status,
2. Soligenous mire – commonly less than 5 ha with limited peat development, occupying springs, flushes, slope hollows and channels,
3. Raised mire – often developed from basin mires, being isolated from groundwater and nutrient poor,
4. Basin mire – formed in topographic hollows and often isolated from groundwater,
5. Valley mire – extending along river valleys and occupying a wide range of base status,
6. Blanket bog – possibly developed from topogenous and soligenous mires and covering a large area, nutrient poor, and
7. Open water transition mire – developed from open water and fed by precipitation.

The system was intended to aid selection of nature reserves, yet is ambiguous and limited in its coverage of wetland sites. More recently and in order to develop a consistent hydrological impact assessment approach, Acreman and Miller (2006) emphasised the importance of water transfer mechanisms. These were combined with landscape location to produce significant types of flatland (upland and lowland), slope, depression and valley-bottom wetland (Figure 1-1).

At a regional scale, classification systems have been structured to particular research outcomes and requirements. In the UK, Lloyd et al. (1993) used examples from East Anglia to produce a classification system with water source and geology as principal controls. The intention was to aid understanding of wetland processes and vulnerabilities. However, relative contributions of surface water and groundwater,

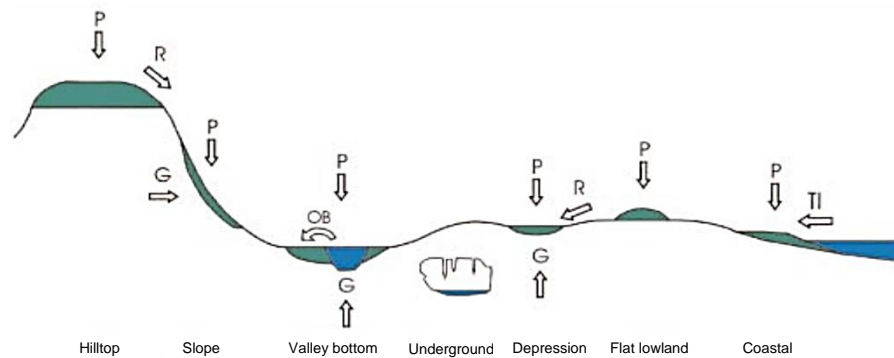


Figure 1-1. Landscape locations and principal water supply mechanisms of wetlands. G, Groundwater; OB, Overbank; P, Precipitation; R, Runoff; TI, Tidal (after Acreman and Miller, 2006)

essential for impact assessment, are not usually known (Acreman et al., 2011a). Gilvear and McInnes (1994) based a hydrological classification of twelve wetland types upon the complete water balance equation, and applied it with success to wetlands in Scotland, although wider applicability is untested. Also in Scotland, SNIFFER (2009) was implemented to assist field identification of wetlands and their hydrological and ecological characteristics for impact assessment. Wheeler et al. (2009) used an ecohydrological framework drawn from a study across central and southern England and Wales. Classification rests on ecological types (water base-richness (pH) and soil fertility categories) and 'WETland water supply MECHANisms' (WETMECs) to identify the main habitats in lowland herbaceous wetlands in England and Wales. Subdivisions further typify wetlands according to particular characteristics, such as groundwater source.

Nonetheless, classification systems may overlook features exclusive to an individual wetland (Lloyd et al., 1993). As part of a study to ascertain sensitivity of fens in East Anglia to groundwater abstraction and pollution, Gilvear et al. (1993; 1997; 1994) illustrated that, although desk study can provide reasonable classifications at regional level, local conditions may render classification for particular sites invalid. Such classification systems do not allow for complexities at higher resolutions and so become inaccurate when applied to discrete wetlands in which hydro-ecological processes differ from the broad typologies used within classification schemes. In an extensive literature review on wetland hydrological function, Bullock and Acreman (1999) concluded that wetlands that appear similar are driven by very different hydrological processes. Hence, almost invariably, data collection is needed to identify the functional role of the wetland.

1.2.2 Global extent

Due to the ambiguities in definition, discussed in the previous section, there is no single and precise figure for the global extent of wetlands, though they are found in all climates and every continent except Antarctica (Mitsch and Gosselink, 2007). Estimates of global wetland area vary considerably by source (Table 1-2). They have been said to cover around 6% of the world's land surface (Maltby and Turner, 1983a), although estimates of global land surface themselves vary widely (Mitsch and Gosselink, 2007). The highest estimate ($12.8 \times 10^6 \text{ km}^2$) arises from the Global Review of Wetland Resources and Priorities for Wetland Inventory (GRoWI) (Finlayson et al., 1999), which adopted the Ramsar definition of wetlands. This includes aquatic ecosystems, such as freshwater lakes, reservoirs, rivers and near-shore marine environments up to 6 m depth, not included in other definitions. Nevertheless, many wetlands are not included or under-represented due to unavailable or partial data (MEA, 2005; Rebelo et al., 2009). At the other end of the scale, studies by Matthews and Fung (1987), who used global digital databases, and Aselmann and Crutzen (1989), who employed regional surveys, contained deficiencies in tropical and subtropical regions (Mitsch and Gosselink, 2007). More recently, the Lehner and Döll (2004) estimate ($8.2\text{-}10.1 \times 10^6 \text{ km}^2$) provided a comprehensive examination through the Global Lakes and Wetlands Database (GLWD), from which the distribution of the world's wetlands is shown in Figure 1-2. The greatest proportion of the world's wetlands occur in the northern boreal regions, where they are influenced by permafrost (OECD, 1996). The remainder are located mainly around the equator in the tropical and subtropical humid regions; of which, around a third occur in arid and sub-arid areas.

Historically, human enterprise has resulted in sweeping degradation and loss of wetland environments (Maltby, 1986; Mitsch and Gosselink, 2007; Thompson and Finlayson, 2001). The current wetland extent likely constitutes about half of what existed prior to human influence (Maltby, 1986). Creation of rich farm and urban land, such as from drainage of sea and coastal marshland in the Netherlands (Idema et al., 1998) or the

Table 1-2. Comparison of global wetland area estimates (after Mitsch and Gosselink, 2007)

Global wetland area estimate ($\times 10^6 \text{ km}^2$)	Source
8.6	Maltby and Turner (1983b)
6.8	Matthews and Fung (1987)
6.9	Aselmann and Crutzen (1989)
12.8	Finlayson et al. (1999)
8.2-10.1	Lehner and Döll (2004)
7.2	Ramsar (2011)

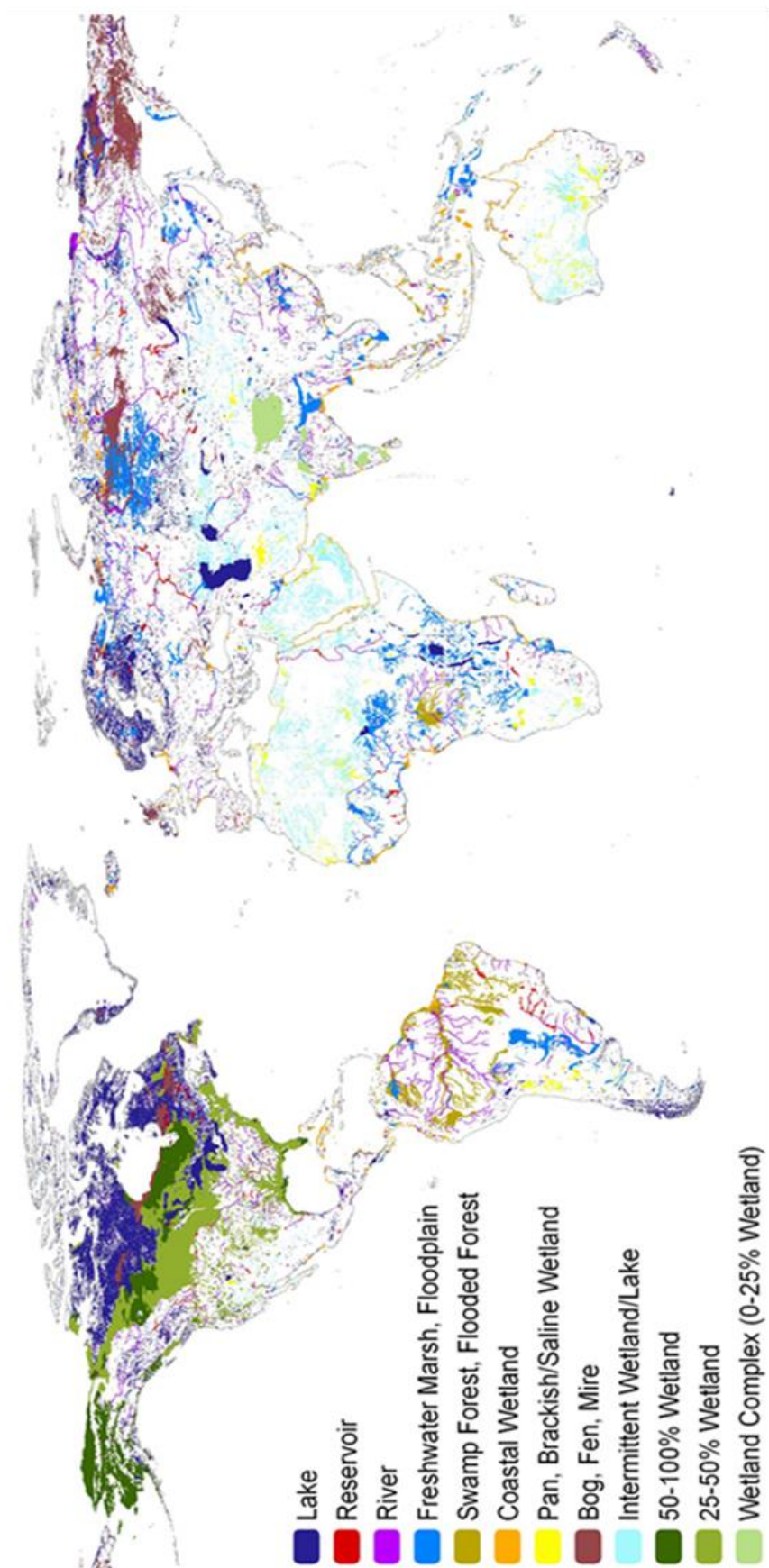


Figure 1-2. Global lakes and wetlands database GLWD (Lehner and Döll, 2004)

lowland fens of England (Darby, 1983), were hailed as necessary enterprises. Drainage was for centuries a practice seen as a progressive and public-spirited endeavour (Baldock, 1984). In Europe, markedly after the Second World War, subsidies encouraged farmers to drain wetlands to increase agricultural output under the EEC's Common Agricultural Policy (Turner and Jones, 2013). In the US, it is estimated that 52 % of wetlands have been lost since European settlement (Dahl and Johnson, 1991). To this day, worldwide wetland degradation has continued (Maltby and Acreman, 2011).

1.2.3 UK wetlands

In the UK there is no national estimate for wetland extent as no definitive register exists. Over the years various databases have focussed on individual wetlands or wetland types, including:

- The Fens of East Anglia (Nature Conservancy Council, 1984)
- Peatlands (RSNC, 1990)
- Estuaries (Buck, 1993)
- Wetlands under the EU Habitats Directive Annex I (Jackson and McLeod, 2000)
- Wetlands under the Wildfowl and Wader Counts (Collier et al., 2005)
- Constructed Wetland Association UK database (Cooper, 2007)
- Wetlands under the UK Biodiversity Action Plan (BAP) (Maddock, 2008)
- National Peat Resource Inventory (NPRI) (JNCC, 2011)

An attempt to collate a number of different datasets by Wetland Vision (2008) (Table 1-3) produced an indicative map of the present-day extent of wetlands in England (Figure 1-3). However, regional and local variability in terms and inconsistencies in data mean that this map is by no means comprehensive and is purposed for indicative use only. Resolution is limited to broad scales, while the analysis was focussed on habitat diversity and designated sites, which cover a fraction of the total current extent of freshwater wetlands (Wetland Vision, 2008). The map under-represents the extent of some wetland habitats, such as fens, blanket bogs and lowland raised mires, and over-estimates the extent of coastal and floodplain grazing marsh. Further developments include an Environment Agency investigation into the possibility of a national digital wetland inventory (Naura, 2006), though this is still pending.

As it has globally, human modification to the landscape has had a significant impact on wetland extent and quality in the UK which is repeated throughout Western Europe. Hollis & Jones (1991) went as far to say that the most common types of wetland in the UK and Europe are either threatened, degraded or lost. Wetlands in the UK are the smaller remnants of once extensive past systems. An indicative map of the historic extent

Table 1-3. Datasets used to define the current indicative extent of wetlands in England (Wetland Vision, 2008)

Dataset	Description	Provider
Agricultural land classification	Agricultural productivity classes	MAGIC; Natural England
BAP priority habitat inventory data	UK Biodiversity Action Plan Habitats including blanket bog, coastal and floodplain grazing marsh, fens, purple moor grass rush pastures, reedbed and wet woodlands	Natural England
UK Lakes Database	Detailed inventory of UK lakes	Environment Agency
Grazing marsh GIS	1999 report data outlining ranking of natural areas for biodiversity	Natural England; Centre for Ecology and Hydrology
HAP Annexe Maps and FenBase Clusters	English Nature Habitat Action Plan Annexe maps	Natural England
NPRI	National Peat Resource Inventory	Natural England
RSPB Reserves	Royal Society for the Protection of Birds Reserve boundaries	RSPB
SSSI units and condition	Sites of Special Scientific Interest primary habitats and condition	Natural England
Wildlife Trust wetland sites	County Wildlife Trust wetland site locations	The Wildlife Trust

for wetlands in the UK based on underlying soil characteristics in presented in Figure 1-4 (Wetland Vision, 2008). Due to fluctuations in area through changes in climate and management, this map does not show where wetlands existed at any chosen point in time but, instead, the maximum former extent.

Agricultural drainage has been evoked as the greatest direct or indirect cause of wetland loss in the UK, followed by mineral extraction, water pollution, road construction, and water abstraction (Gilman, 1994). In the south eastern lowlands, as elsewhere in the UK, agricultural reclamation and groundwater abstraction have led to the loss of large areas of groundwater dependent wetlands (Williams, 1990), such as Redgrave and Lopham Fen, Suffolk (Gilvear et al., 1994; Harding, 1993). Estimates have placed the area of fen in England at 3400 km² in 1637, compared with 10 km² today (McBride et al., 2011).

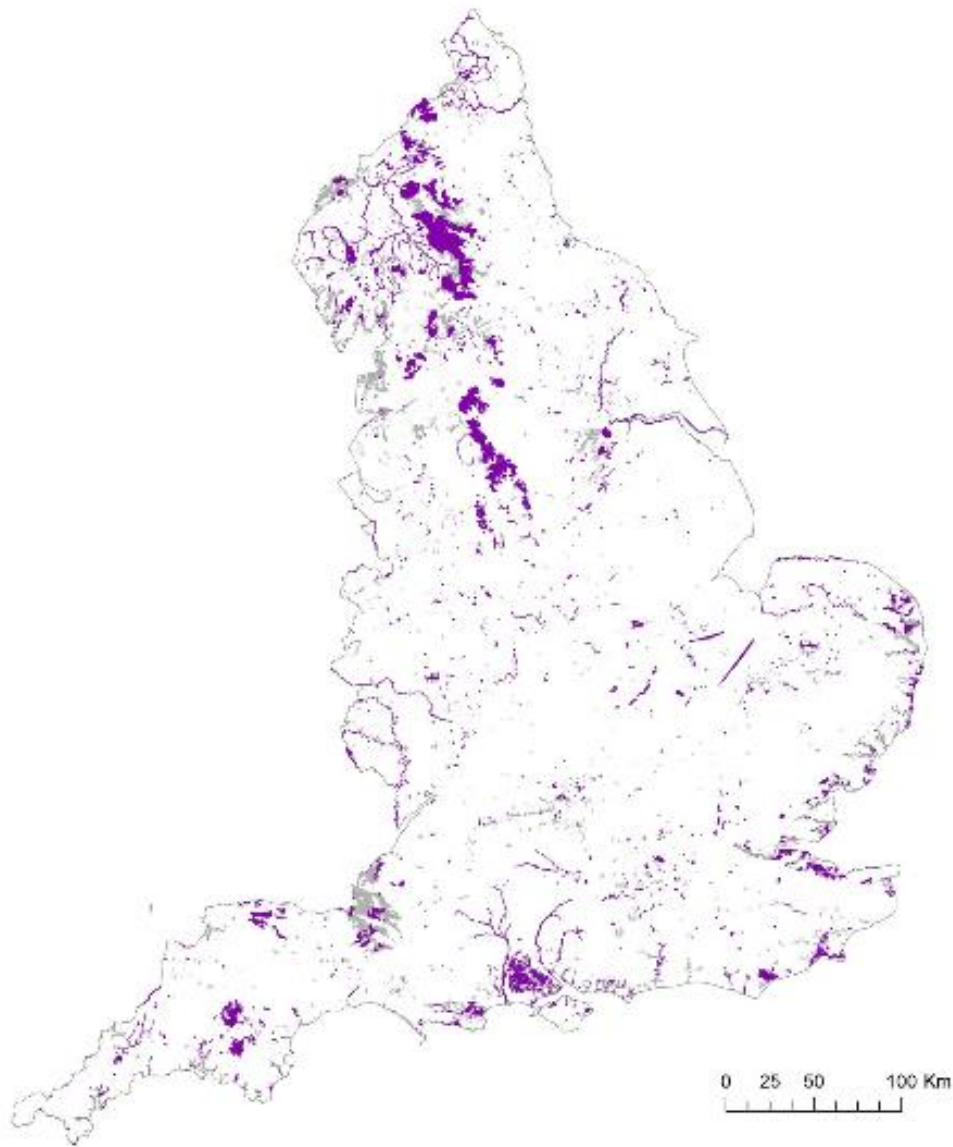


Figure 1-3. The current indicative extent of wetlands in England (Wetland Vision, 2008)

1.2.4 Wetland values and services

Wetlands are now widely considered to provide invaluable environmental, cultural and economic functions and services (Acreman et al., 2011b), although once they were regarded as wastelands (Maltby, 1986). Awareness of the detrimental impact of drainage and other activities leading to wetland loss and degradation (Armentano, 1980; Gambolati et al., 2006; Waltham, 2000) has risen along with the socio-economic significance of wetlands (Maltby, 2009). Historically, wetlands have contributed towards global biodiversity through their role in the evolution of many species (Gopal, 2009), a role that may even include human beings through an increased brain capacity enabled by the provision of a diet of fish and the fatty acids they contain (Morgan, 1982). Human

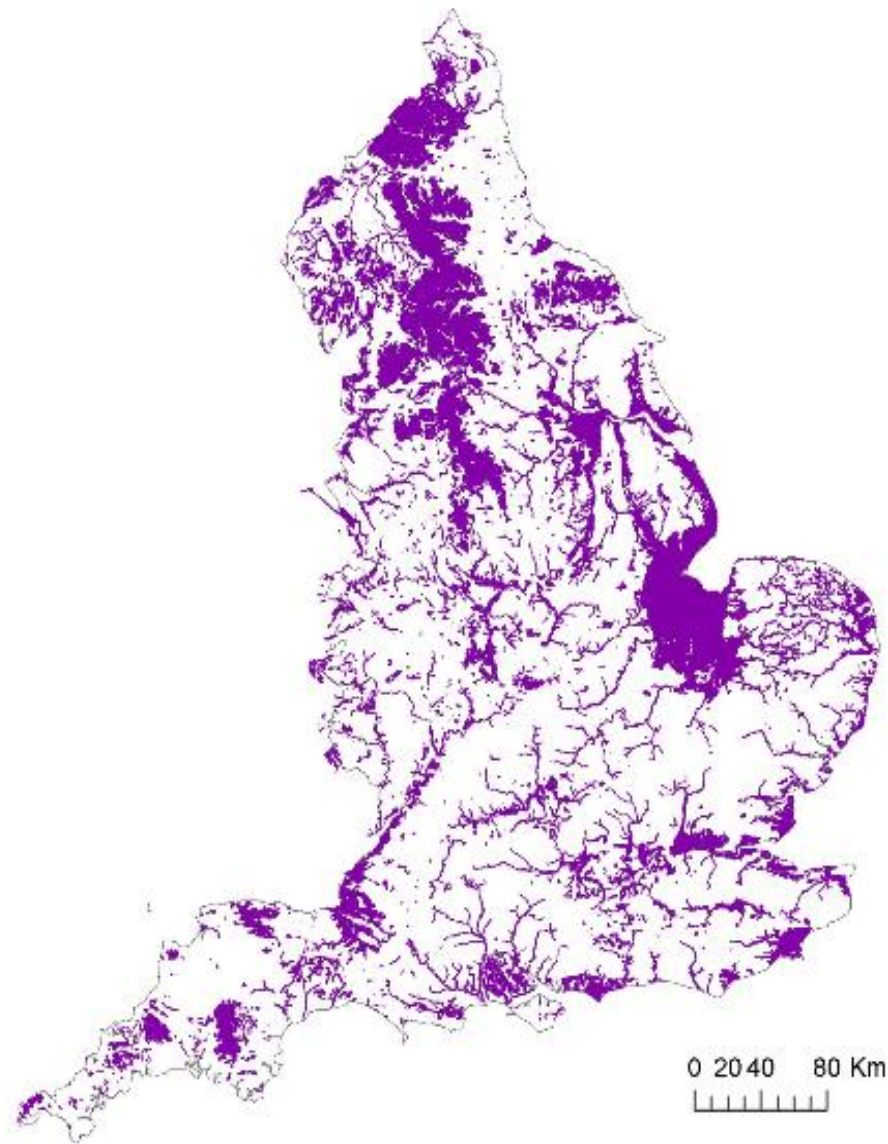


Figure 1-4. The historical maximum indicative extent of wetlands in England (Wetland Vision, 2008)

civilisation is itself founded upon prehistoric communities which occupied the wetland margins of rivers, lakes and seas (Coles and Coles, 1989; Dolukhanov, 1992; Mitsch and Gosselink, 2007). Wetlands continue to directly support people, their communities and livelihoods around the world, contributing to human well-being and poverty alleviation (MEA, 2005; Ramsar, 2011). They provide more benefits to urban populations than any other ecosystem (McInnes, 2014). Accruing descriptions such as ‘biological supermarkets’ and ‘kidneys of the landscape’ (Mitsch and Gosselink, 2007), rich in the rhetoric of value and good health, the perception of wetlands in the past 30-40 years has shifted to one of appreciation.

This positive shift in view may be envisaged within the concept of ecosystem services (Figure 1-5). These are natural assets (Barbier, 2011) produced by the environment and used by mankind, or “the benefits people obtain from ecosystems” as described by the Millennium Ecosystem Assessment (MEA, 2005). They aid social and cultural well-being (Fisher et al., 2009) and hold high economic value (Barbier et al., 1997; Fisher and Turner, 2008). Classified under the categories of provisioning, regulating, cultural and supporting (MEA, 2005), they are normally interpreted as what in economics would be “goods and services” (Barbier, 2007).

In the context of wetland research, ecosystem services have their roots in concepts of ecosystem functioning and resulting human values (Maltby, 1986). Some examples of the specific services wetlands provide through their underlying functions are listed in Table 1-4, and, although not exhaustive, relevant studies are also provided. Those that have easily identified economic value include food production, hay for cattle fodder, peat for fuel and horticulture, reeds for thatching, potable water supply, environments for tourism and recreation. Less quantifiable are the services and functions which often form the case for wetland conservation, including flood and coastal protection, groundwater

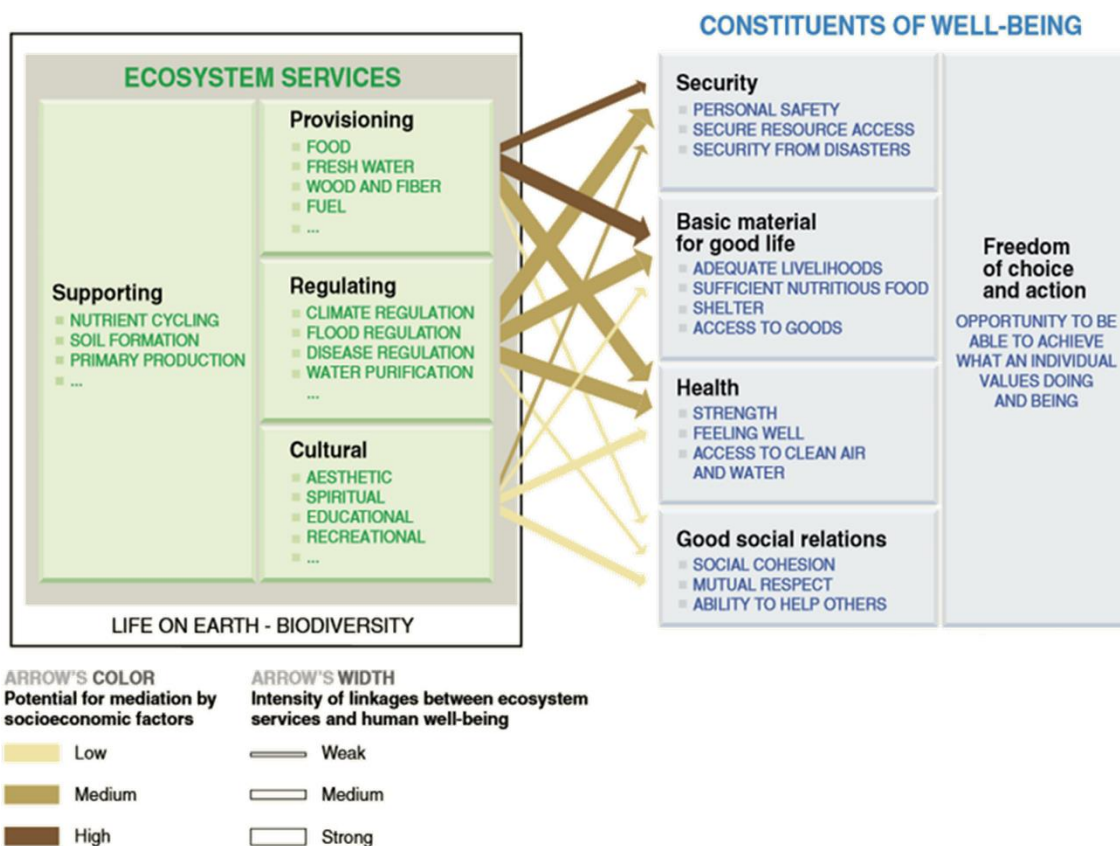


Figure 1-5. Interactions between ecosystem services and human well-being (MEA, 2005)

Table 1-4. Examples of wetland services, functions and relevant studies

Service	Function	Example Studies
Flood protection	Dispersal / storage of flood flows	Acreman (2003); Williams (1990)
Coastal protection	Flow regulation and control	Brouwer and van Ek (2004)
	Wave attenuation / dissipation, wind buffer	Badola and Hussain (2005); Costanza et al. (2008); Das and Vincent (2009); King and Lester (1995)
Erosion control	Sediment deposition / retention	Johnston et al. (1984); Karr and Schlosser (1978); Newall and Hughes (1995); Sathirathai and Barbier (2001); Wheeler and Shaw (1995)
Carbon storage	Bank stabilisation	Gregory (1992)
	Carbon sequestration and flux, peat formation	Gorham (1991); Heathwaite (1993); Immirzi et al. (1992); Jenkins et al. (2010)
Conservation	Habitat provision	Brouwer and Bateman (2005); Do and Bennett (2009); Gopal and Masing (1990); Jenkins et al. (2010); Merritt (1994); Ron and Padilla (1999)
Water supply	Groundwater recharge / discharge	Acharya and Barbier (2000); Lloyd and Tellam (1995)
Water quality	Nutrient source / sink	Burt et al. (1993); Fisher and Acreman (2004); Haycock and Burt (1993); Lee et al. (1975); Lowrance et al. (1984); Yang et al. (2008)
	Immobilisation of environmental contaminants	Hellawell (2012) Allinson et al. (2000); Ross (1995)
	Urban runoff sink	Hollis (1979); Hollis and Ovenden (1988); Kansiime and Nalubega (1999)
Produce	Arable and pastoral environment for hay, peat, forestry, fish, reeds	Barbier (2007); Barbier (2011); Islam and Braden (2006); Maltby (1986); Maltby (1991); Smith (2007)
Societal values	Recreation, tourism, education and aesthetic	Hook et al. (2012) Andrews and Kinsman (1990); Bateman and Langford (1997)
	Archaeology	Coles (1990)
	Paleoenvironmental record	Heathwaite and Gottlich (1993)

recharge, erosion control and water quality improvement (Maltby, 1991). Many of these functions further combine to produce biodiversity and landscape, heritage and more esoteric attachments. The survival of many species is dependent on wetlands and their intrinsic functioning. In the UK alone it has been estimated that over 3500 species of invertebrates, 150 species of aquatic plants, 22 species of duck and 33 species of wading birds rely on wetland ecosystems (Merritt, 1994).

1.3 Groundwater dependent wetlands

1.3.1 Hydrology of groundwater dependent wetlands

The hydrological regime is the principal factor in the establishment and maintenance of specific wetlands and wetland processes (Mitsch and Gosselink, 2007). However, as seen in Section 1.2.1, interactions between topography, climate, geology and human influence are also important. In tropical and temperate areas it is common for wetlands to have multiple water sources, while in more arid regions, groundwater may be the only significant input (Lloyd and Tellam, 1995). In the UK, multiple source wetlands are frequently set within a complex geological environment, often composed of superficial deposits, which lead to intricate groundwater/surface water interactions. These wetlands are typically small in scale compared to the surface water and groundwater catchments that contribute to their existence. The importance of local conditions becomes especially apparent when wetlands are found in areas where the hydrogeological conditions might be expected to limit wetland development. For example, despite the permeability of the bedrock across the freely draining chalk of south-east England, wetlands are widely established along river valley bottoms. These areas have a long history of cultivation as water meadows (Everard, 2005). Yet, water fluxes and ecological processes remain poorly understood.

1.3.1.1 Wetland water balance

Interactions between physical components and hydrology will vary between wetlands (Table 1-5). These components and the factors they influence control the relative importance of the terms in the water balance as well as their temporal variations. In turn this impacts wetland function and provision of ecosystem services (Gilman, 1994; Hollis and Thompson, 1998), which were detailed in Section 1.2.3. Management of a wetland requires knowledge of the water balance and its component interactions. Attention may be drawn to processes requiring further investigation through noticeable inequalities between total inputs and total outputs (Acreman and Miller, 2006). However, accurately quantifying a water balance can prove challenging and often becomes iterative. Complexities, such as the association of the wetland to an underlying aquifer, must be taken into account. The unfeasibility of precise measurement of water transfer rates, along with errors in calculation or the equation terms (Dooge, 1975), lead to uncertainty. Although this may be presented explicitly, if the imbalance between inputs and outputs is greater than the uncertainty of the measurements, the conceptual understanding of wetland hydrological functioning will require modification.

Table 1-5. Summary of factors controlling wetland hydrology and function (after Lloyd et al., 1993)

Component	Controlling factor
Topography	Catchment area size, rate and energy of flow.
Climate	Groundwater recharge, surface-water flow, precipitation, evapotranspiration, role of wetland in hydrological regime.
Vegetation	Leaf area index (LAI), interception, evapotranspiration.
Geology	Groundwater flow, hydrochemistry, proportions of groundwater and surface water, springs, hydraulic continuity with aquifer.
Human	Land drainage, abstraction, pollution, eutrophication, urbanisation.

Precipitation and evapotranspiration are components of the water balance in all UK wetlands. The distribution of precipitation (Figure 1-6) and its seasonality provide the

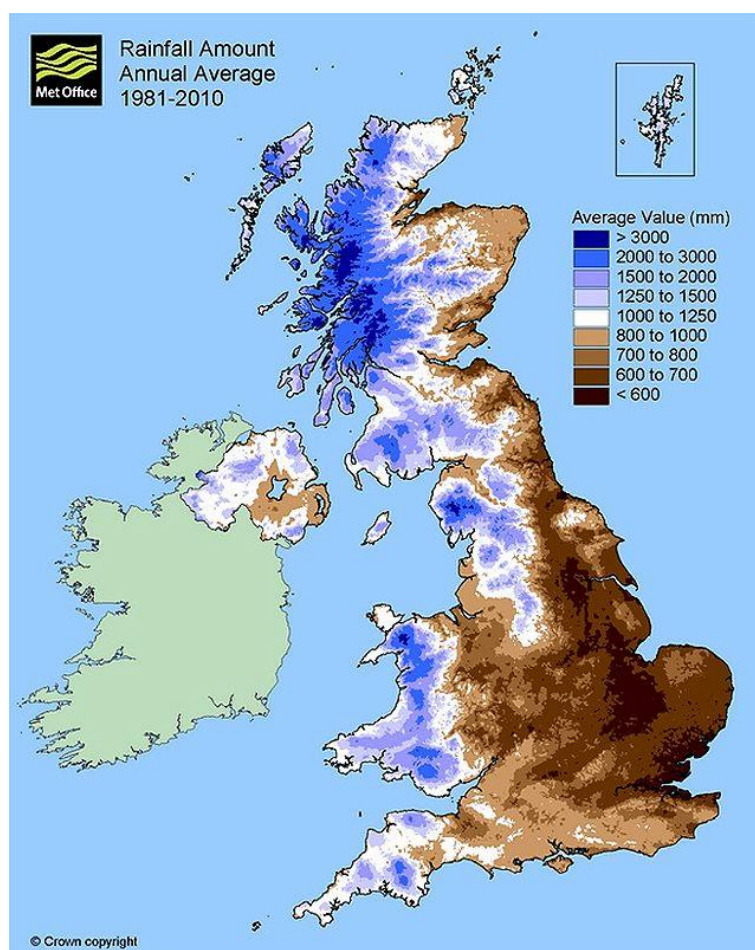


Figure 1-6. Annual mean precipitation for the UK: 1981-2010 © Crown copyright 2016 Met Office

climatic background for wetland development (Gilman, 1994). In the west of the UK, upland areas of Wales and the Lake District annual average rainfall exceeds 3000 mm, while much of eastern England receives less than 700 mm per year. The high rainfall in the western uplands, which is greatest during winter months, ensures the development of upland mires. In central and eastern England, the effects of drought may be relieved by convective storms in the summer months. However, wetland water levels are mainly restored through winter by runoff and groundwater flow. This limits wetland development to areas with larger catchments. Wetland development by precipitation alone is limited to the east and higher altitudes.

Evapotranspiration rates are proportional to the difference between the vapour pressure at the water (or leaf) surface and the vapour pressure in the air. Evapotranspiration rates in wetlands are rarely limited by the soil moisture deficit, unless highly seasonal, and are driven by meteorological conditions such as solar radiation and temperature. In the Bells Creek catchment of the North Kent Marshes, southeast England, annual evapotranspiration has been reported to account for between 63% to 87% of the outflows for the catchment (Hollis and Thompson, 1998). The ratio of evapotranspiration to potential open water evaporation is important to understanding the role of vegetation in the water balance (Crundwell, 1986). Dependent on the plant species, growth, climate and density, evapotranspiration rates may exceed or fall below those of evaporation from open water (Gilman, 1994). Temporal evapotranspiration rates are controlled by seasonal vegetation growth and changes in cover and density (Baker et al., 2009).

Surface water flow into and within wetlands may arise from overland flow, channelised stream flow, or overbank flow from adjacent waterbodies (Hollis and Thompson, 1998). Overland flow can result either from infiltration-excess overland flow, when rainfall intensity exceeds infiltration (Betson, 1964; Horton, 1945), or saturation-excess overland flow, which occurs at lower rainfall intensities and is produced when water rises to the surface as the soil is saturated (Hewlett and Hibbert, 1967). Riparian wetlands, which occur along the adjacent floodplains of a river or stream will be periodically flooded by overbank flow. Characteristics of flood frequency, duration and magnitude, which control wetland water levels are themselves controlled by the river regime which is in turn the result of interactions of precipitation and evapotranspiration over the wider catchment (Baker et al., 2009).

Once water has infiltrated, subsurface flow (throughflow) may occur through micropore spaces in the soil matrix, macropore networks of cracks and root channels (Beven and Germann, 1982), or larger cavities known as soil pipes (Jones, 2010). Flow through

macropores and micropores can vary by several orders of magnitude depending on the presence of different pathways at different scale, as found by Bromley et al. (2004) at Thorne Moor, North Yorkshire. Features such as abandoned infilled ditches, root holes and localized woody material, can cause peat hydraulic conductivity to be heterogeneous, and to increase with scale. Dependent on soil type, the influence of macropores on flow may vary temporally due to shrinkage or swelling with saturation (Thompson et al., 2004).

Groundwater may contribute a significant proportion of the wetland water balance (Bravo et al., 2002; Krause and Bronstert, 2005). For example, groundwater inflows have been shown to account for up to 90% of water inputs to Badley Moor Fen, East Anglia, UK (Gilvear et al., 1993). Groundwater flow is influenced by topographical, geological and climatic factors (Sophocleous, 2002; Winter, 1999). Differences in topography are often reflected in groundwater levels. That is, assuming a spatially uniform precipitation and infiltration rate over an undulating surface, groundwater flow will be regulated by a water table surface that follows the ground surface, albeit in a subdued way (Hubbert, 1940). The rate and direction of groundwater flow will also be controlled geologically by the distribution of hydraulic conductivity through the subsurface, and climatically by the intensity and distribution of precipitation, which, after moderation by evapotranspiration, drives recharge. Groundwater flow systems may be local, discharging to a stream or pond, regional, discharging to major rivers, lakes or oceans, or occupying an intermediate position between the two (Toth, 1963) (Figure 1-7). The interaction of wetlands with groundwater is governed by the position of the wetland with respect to different-scale groundwater flow systems, geological controls on seepage distribution, and the magnitude and distribution of precipitation and evapotranspiration (Winter, 1999).

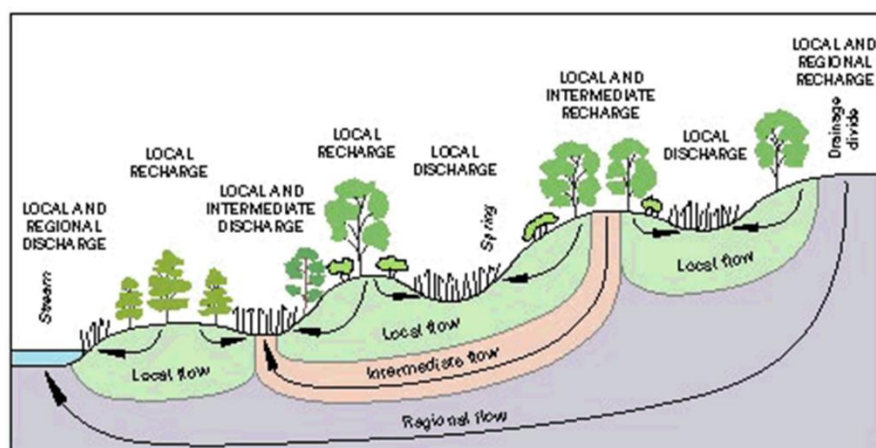


Figure 1-7. The different scales of groundwater flow systems (Carter, 1996)

1.3.1.2 Wetland interactions with groundwater

Individual wetlands may differ significantly in their interactions with groundwater, despite being geographically close. It is important to consider local complexity, as local geology can create varying flow pathways. For example, in eastern England the three Breckland Meres (small lakes), Langmere, Ringmere and Fenmere, are visually similar and geographically close, within 2 km of one another, yet hydrologically different (Acreman and Jose, 2000) (Figure 1-8). Langmere is in direct contact with the underlying Chalk aquifer and groundwater fluctuations control its hydrological regime. Ringmere is partially separated from the aquifer by an aquitard layer of organic matter, yet is still predominantly groundwater controlled. Fenmere, on the other hand, is isolated from the aquifer by an aquiclude clay layer, with water levels controlled by precipitation and evapotranspiration.

Although many different types of wetland have been classified (Section 1.2.1), there are two fundamental settings for groundwater dependent wetlands in the UK (Lloyd and Tellam, 1995). These are, firstly, groundwater fed wetlands connected to a surface water body (i.e. a stream, river or lake) and, secondly, groundwater fed wetlands independent of surface water bodies. In the first setting, groundwater contributions are head controlled

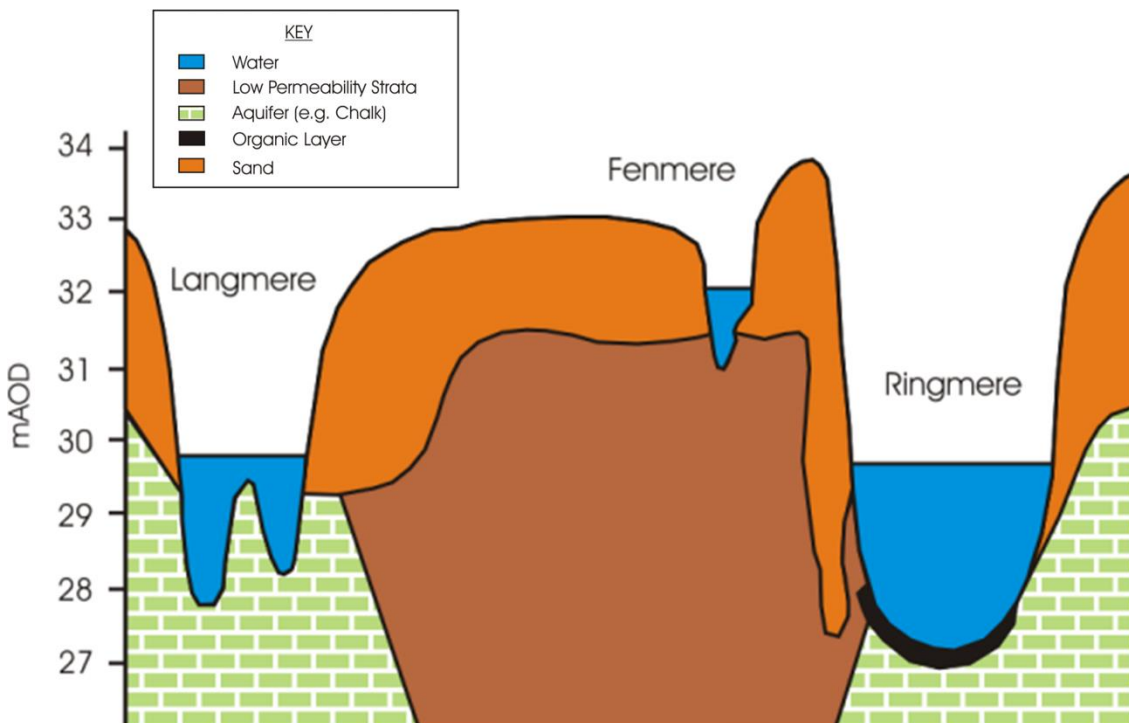


Figure 1-8. Geological cross-section through the Breckland Meres, UK (after Acreman and Jose, 2000)

by the adjacent water body. Other sources aside from groundwater may be equally or more significant. In the second setting, groundwater levels are predominantly topographically controlled, formed at the intersection of the groundwater head distribution with a topographic depression. Figure 1-9 illustrates a schematic of a typical UK wetland water balance with a groundwater element.

Valley bottom wetlands, such as those of the UK Chalk lowlands, correspond to the first setting. The water balance may incorporate significant elements of both surface water and groundwater (Bravo et al., 2002; Krause and Bronstert, 2005). Generalised cross-sectional representations of water transfer mechanisms for groundwater fed valley bottom wetlands are shown in Figure 1-10 (Acreman and Miller, 2006). Inputs into the wetland are dominated by groundwater discharge with overbank flow from an adjacent watercourse, supplemented by runoff from adjacent higher ground and direct precipitation. Outputs include groundwater recharge when water table is low, drainage to the watercourse when levels in the latter are lower than those in the wetland, surface flows to the watercourse and evaporation. Differences arise from the presence or absence of a low permeability layer separating the aquifer from the wetland, which can restrict groundwater flow.

Wheeler et al. (2009) identified similar processes in two wetland categories, “groundwater fed floodplains” and “groundwater bottoms”, corresponding to WETMECs 7 and 9 of their classification system (see Section 1.2.1). The first, with examples in Bransbury Common, Chilbolton Common and Greywell Fen, Hampshire, and Chippenham Fen, Cambridgeshire, typifies undrained floodplains of groundwater-fed

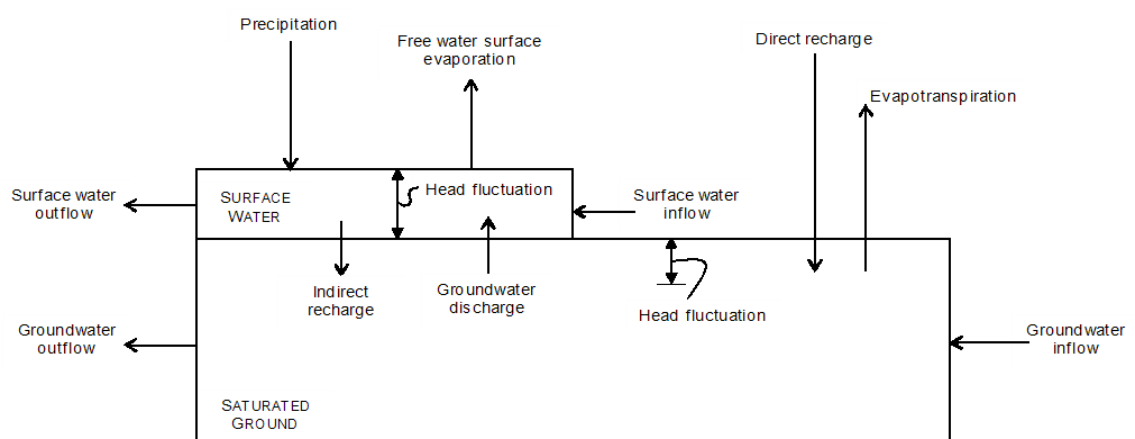


Figure 1-9. Generalised water balance for a surface and groundwater-fed wetland (after Lloyd and Tellam, 1995)

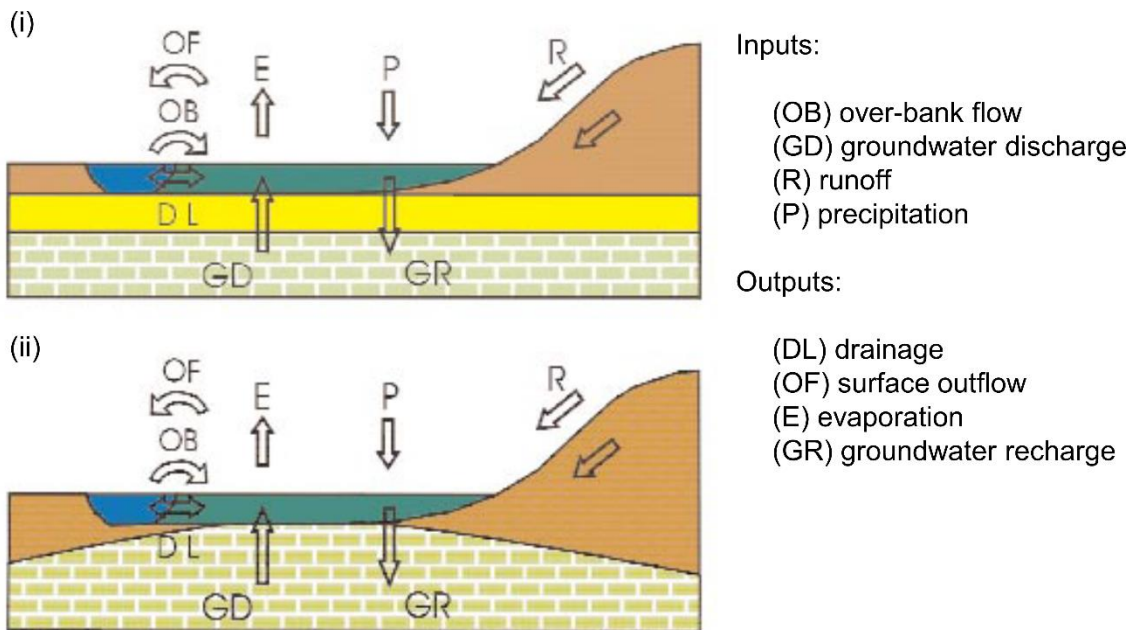


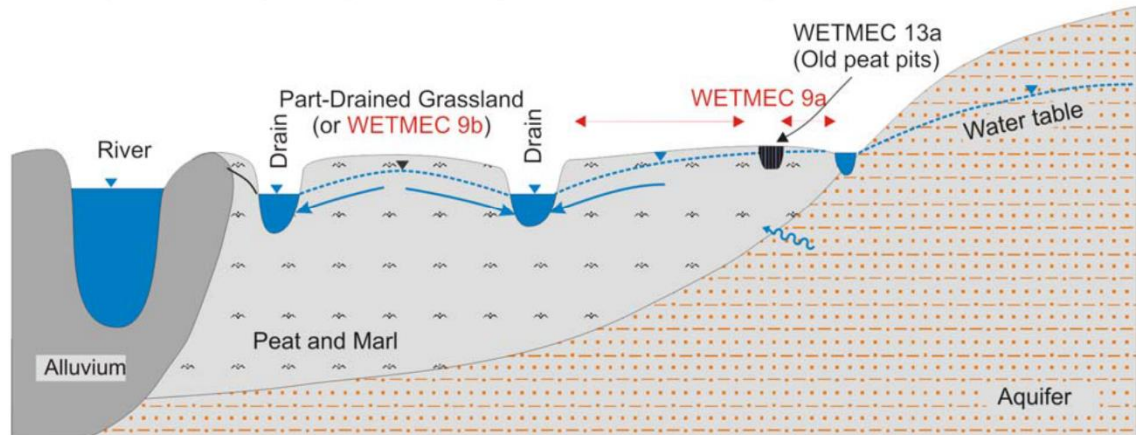
Figure 1-10. Water transfer mechanisms for (i) a surface and groundwater-fed valley bottom wetland where a low permeability layer may limit groundwater-surface water interaction, and (ii) a groundwater-fed valley bottom wetland in direct contact with the underlying aquifer (Acreman and Miller, 2006).

ivers over heterogeneous and slowly permeable alluvial deposits. Both the river and wetland are fed by groundwater, with river levels in equilibrium with the piezometric head of the aquifer. Groundwater levels in the floodplain are equivalent to river levels, although areas of the floodplain may be seasonally dry. However, the category is ill-defined, and without schematic. The second corresponds to floodplain margins and valleyhead basins (Figure 1-11), with examples in Bransbury Common, Hampshire, Blo' Norton and Thelnetham Fens, Cavenham Poor's Fen, Hopton Fen, and Pakenham Meadows, Suffolk, plus East Ruston Common, Limpenhoe Meadows, Redgrave and Lopham Fens in Norfolk. Groundwater discharge is the main water source, although the water table is regularly or permanently below the ground surface.

Groundwater/surface-water interaction can vary considerably over time, dependent on previous recharge. For example, in Spain, Las Tablas de Daimiel wetlands are fed by discharge from the underlying aquifer and Guadiana river when groundwater levels are high; though when groundwater levels are low, the wetlands provide recharge to the aquifer (Llamas, 1989). Hunt et al. (1999) showed that periodic high groundwater discharge to natural and constructed wetlands was correlated with high water tables. Wetlands with larger and more continuous groundwater inputs had less variability in their hydrographs. Additionally, groundwater flow to wetlands may be spatially heterogeneous

WETMEC 9a: 'Wet' Groundwater Bottoms
(e.g. Theltham Fen)

- valley bottom is fed by groundwater
- highest water tables occur along upland margin, where hollows (old peat workings) can support locally very wet conditions
- partial drained of valley bottom and low river levels result in lower water tables away from the upland margin, and in some cases these areas support farmland
- marl layers within the peat may form local aquitards, but are not laterally extensive



WETMEC 9a: 'Wet' Groundwater Bottoms
(e.g. Poplar Farm Meadows)

- valley bottom is pump-drained and no longer normally receives episodic river flooding
- margin is fed by groundwater, where there appears to be localised upflow
- alluvial clays close to the river help to confine the aquifer locally and, when drained, provide a firm surface suitable for livestock

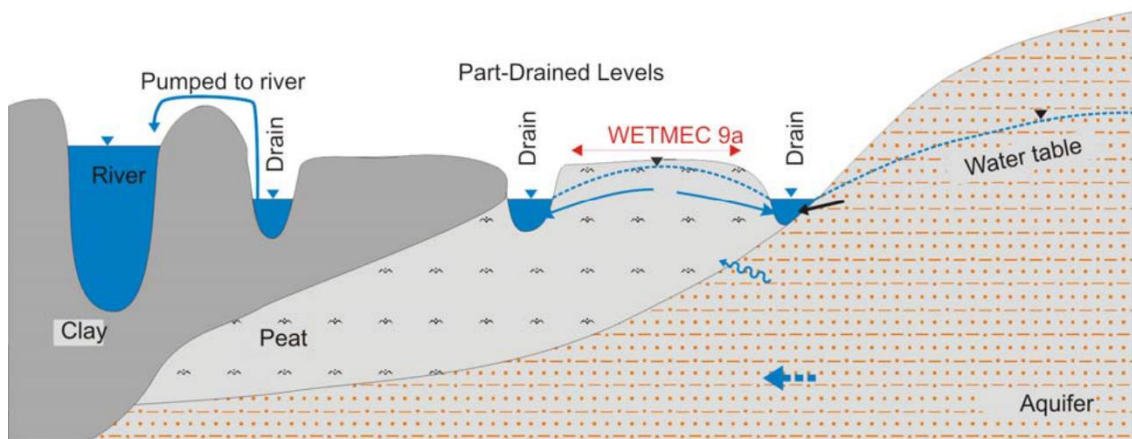


Figure 1-11. Schematic cross-sections of groundwater-fed valley bottom wetlands (Wheeler et al., 2009)

(Hunt et al., 1996; Lowry et al., 2007). Prairie-pothole wetlands in Wisconsin, US, have been shown to contain areas of both recharge and discharge (Hunt et al., 1996), with analogous findings in North Dakota (Winter and Rosenberry, 1995). Likewise in Wisconsin, discrete zones of groundwater discharge have also been shown in a stream within a peat-dominated wetland (Lowry et al., 2007). The corollary being that point

measurements of groundwater flux, if extrapolated uniformly over a wetland, could miss discrete areas of discharge or recharge.

1.3.2 Hydroecology of groundwater dependent wetlands

Hydrological conditions govern wetland ecology (Mitsch and Gosselink, 2007), being a principal control on vegetation (Baldwin et al., 2001; Wheeler et al., 2009), fauna (Ausden et al., 2001; McMenamin et al., 2008) and biogeochemical cycling (Lischeid et al., 2007; McClain et al., 2003). Differences in the water supply mechanisms and hydrological regime will influence the abiotic factors, such as nutrient supply, that control species presence and distribution (MEA, 2005). The distinct chemical properties of groundwater (Acreman and Miller, 2006), which may form a large proportion of the water balance (see Section 1.3.1.1), support different floral and faunal communities to those fed by surface water or precipitation alone (Klijn and Witte, 1999). Hence, groundwater dependent wetlands provide habitats for diverse species important to conservation (Fojt, 1994) and are listed as priority habitats under the European Habitats Directive (EEC, 1992).

1.3.2.1 Effect of the hydrological regime on wetland ecology

The hydrological regime of a wetland affects numerous abiotic factors, such as nutrient availability and anaerobic soils, which in turn determine the biotic components (MEA, 2005). Different flows and water levels support a range of ecological functions and services (Table 1-6). Changes in the depth, duration, frequency, magnitude and timing of water supply has significant implications for the type of plants that will grow in a wetland (Wheeler and Shaw, 1995). The magnitude, frequency, duration and timing of changes in flow and water levels influence biotic communities and ecosystem processes through their effects on other primary regulators (Batzler and Sharitz, 2014; Langhans and Tockner, 2006). Alterations to the wetland hydrological regime may therefore change the assemblage of plants and animals present, dependent on the magnitude of change. The consequence is that wetlands are often highly vulnerable and sensitive to environmental change.

In channels, elements of the flow regime, including flooding, mean flows and low flows, are important to sustaining the ecology (Poff et al., 1997). Changes in river flow regime will cause an ecological response due to a direct relationship between physical habitat and flow (Beecher et al., 1993; Bovee, 1982; Bovee et al., 1998; Cavendish and Duncan, 1986). Flow in this sense is used as proxy for water depth and velocity, as these provide physical habitat for plants, invertebrates and fish through interactions between

Table 1-6. Different flows in support of wetland ecology and services (MEA, , 2005)

Flow component	Ecological role
Low (base) flows	Provide adequate habitat space for aquatic organisms
Normal level:	Maintain suitable chemical conditions, including dissolved oxygen
	Maintain water table levels in floodplain and plant soil moisture
	Provide drinking water for terrestrial animals
	Keep fish and amphibian eggs suspended
	Enable passage of fish to feeding and spawning areas
	Support hyporheic organisms (living in saturated sediments)
Low (base) flows	Enable recruitment of certain floodplain plants
Drought level:	Purge invasive, introduced species from aquatic and riparian communities
	Concentrate prey into limited areas to the benefit of predators
Higher flows (small flood pulses)	Shape physical character of river channel, including availability and heterogeneity of different biotopes (such as riffles, pools) and microhabitats
	Restore normal water quality after prolonged low flows, flushing away waste products, pollutants, and proliferations of nuisance algae
	Maintain suitable salinity conditions in estuaries
	Prevent encroachment of riparian vegetation into the channel
	Aerate eggs in spawning gravels, prevent siltation of cobble interstices
	Determine size of river bed substrata (sand, gravel, cobble, boulder)
Large floods	Provide fish migration and spawning cues
	Provide new feeding opportunities for fish and waterfowl
	Recharge floodplain water table
	Maintain diversity in floodplain forest types through prolonged inundation
	Control distribution and abundance of plants on floodplain
	Trigger new phases of life cycles (such as insects)
	Enable fish to spawn on floodplain, provide nursery area for juveniles
	Deposit nutrients on floodplain
	Maintain balance of species in aquatic and riparian communities
	Create sites for recruitment of colonizing plants
	Shape physical character and habitats of river channels and floodplain
	Deposit substrata (gravel, cobble) in spawning areas
	Flush organic materials (food) and woody debris (habitat structures) into channel
	Purge invasive, introduced species from aquatic and riparian communities
	Disburse seeds and fruits of riparian plants
	Drive lateral movement of river channel, forming new habitats (secondary channels, oxbow lakes)
	Provide plant seedlings with prolonged access to soil moisture
	Drive floodplain productivity

the flow rate and channel morphology (Gallagher and Gard, 1999; Gore et al., 1998; Jowett et al., 2005; Jowett, 1992). Habitat Suitability Indices (HSIs) may be used to convert hydraulic descriptions into a measure of available physical habitat (Dunbar et al., 2001) (Figure 1-12). These rely on relationships between hydraulic variables and habitat use, dependent on many factors, including but not limited to species, size and life stage, temperature, season, light levels and food availability. The amount of physical habitat has been shown to be an important control on trout abundance (Jowett, 1992), benthic community diversity (Gore et al., 1998), and the spawning density of

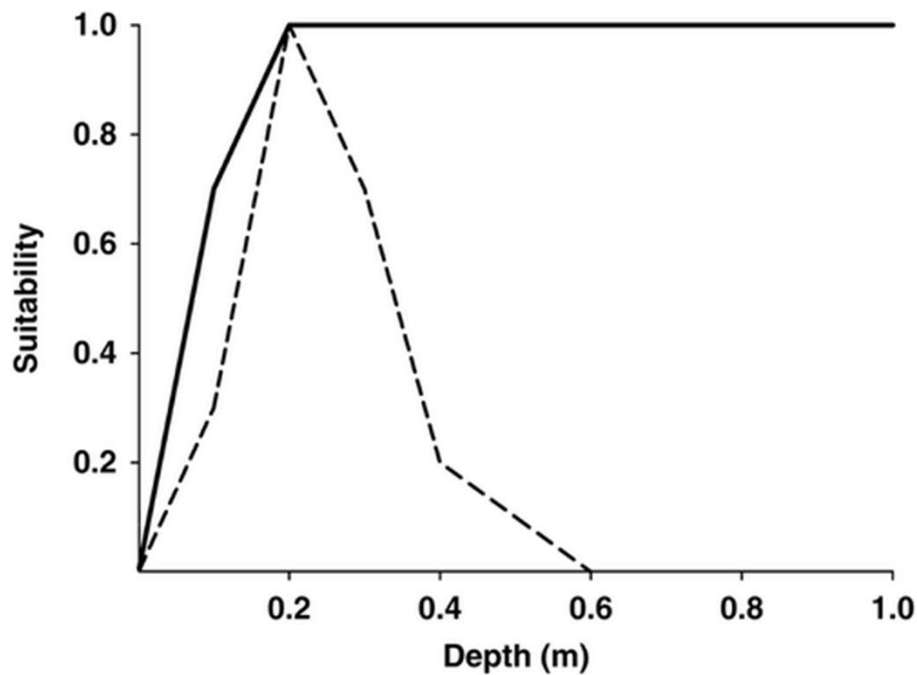


Figure 1-12. Example HSIs for the depth variable (after Conallin et al., 2010). Thick line: large range of conditions result in near optimum habitat conditions, Dashed line: low range of conditions result in near optimum habitat conditions

salmon (Gallagher and Gard, 1999). In the UK, greater flow variability in upland catchments is reflected in higher habitat variability compared to chalk lowland catchments (Dunbar et al., 1996).

In wetlands, water table level regime is a dominant control on plant communities (Silvertown et al., 1999), with the length and depth of inundation affecting vegetation (Baldwin et al., 2001; van der Valk, 2005). Large changes in the range of water depth (over 1.5 m) will lead to a change in species composition for an area, whilst low changes in the range (less than 0.5 m) will cause a change in relative species abundance. In a range of experiments, Steven and Toner (2004) found continuous flooding of coastal plain depression wetlands yielded the lowest vegetation species richness out of vegetation types ranging from open-water ponds and emergent marshes to closed forests. Non-flooded but moist wetlands produced the highest species richness, and ephemeral intermediate. Seedling recruitment and vegetative growth was inhibited in most species by high water level. Additionally, flooding during the first half of the growth season reduced species recruitment. In the UK, the preferred water levels and depths to groundwater for wetland plants and communities have been well documented (Elkington et al., 1991; Gowing et al., 2002; Newbold and Mountford, 1997; Wheeler et al., 2004; Wheeler et al., 2009). Hydrological requirements have been primarily assessed for national vegetation classification (NVC) communities (Rodwell, 1994-2000) that

contribute to features designated as being of importance under the European Habitats Directive (EEC, 1992), and that are found in the lowlands of eastern and southeast England. Figure 1-13 presents example diagrams of water level zones, which illustrate mean monthly water table requirements for selected communities; M16 wet heath/degraded mires and MG8 floodplain margins. Green areas indicate desirable conditions, amber represents tolerable conditions should the water table fall within these zones for limited periods, and red is indicative of intolerable conditions. Wet heath vegetation (M16) requires periodically waterlogged soils, especially in the winter (Elkington et al., 1991), while floodplain margin vegetation (MG8) needs continuously shallow water level depths.

Modifications to a wetland's hydrological regime may also induce changes in animal species distribution. Of especial focus are the impacts on wading birds and their

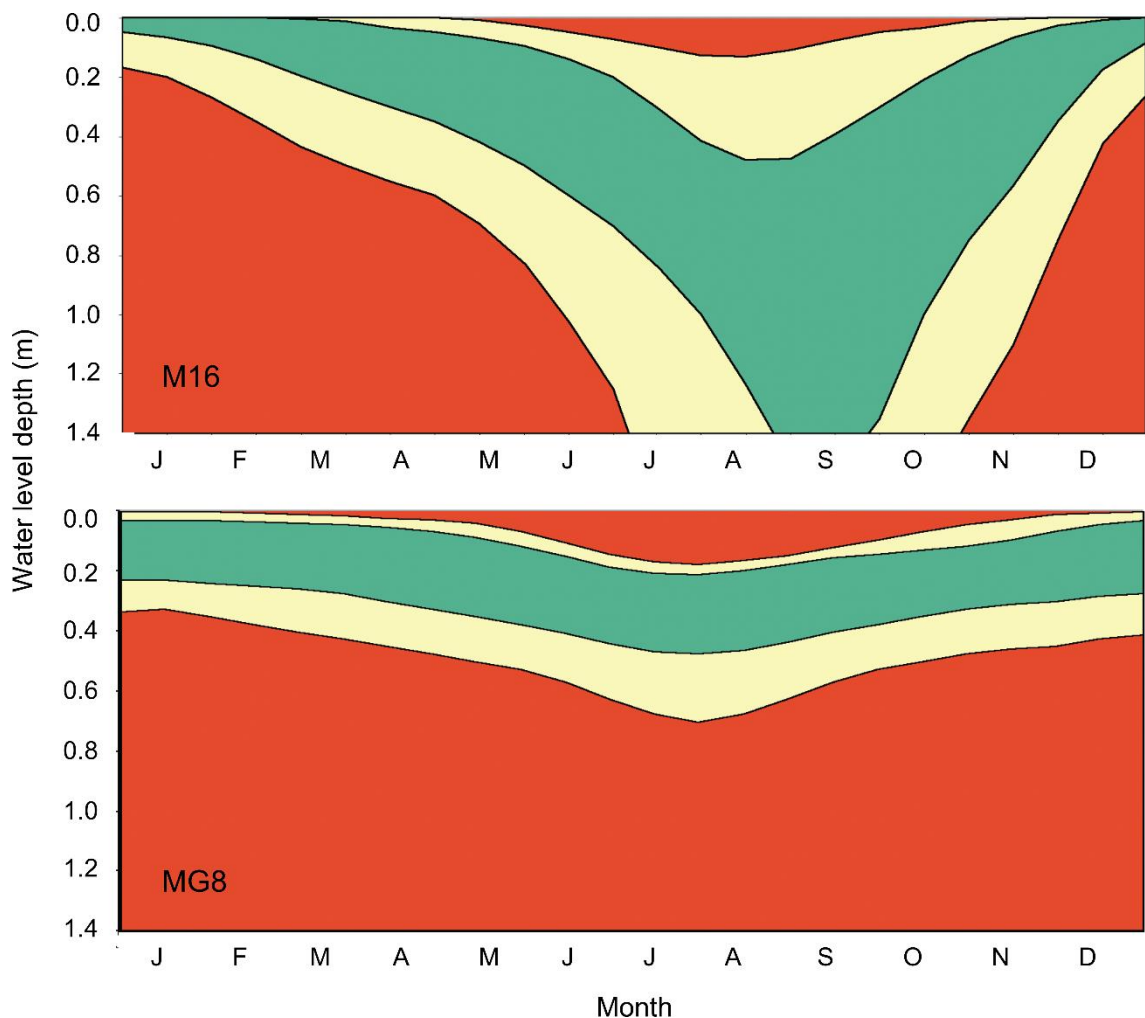


Figure 1-13. Water level requirements for M16 wet heath and MG8 floodplain margins vegetation communities (after Wheeler et al., 2004)

macroinvertebrate prey. Declines in lapwing (*Vanellus vanellus*), snipe (*Gallinago gallinago*), redshank (*Tringa totanus*) and black-tailed godwit (*Limosa limosa*) (Figure 1-14) have been linked to losses in lowland wet grassland in the UK and Europe (Ausden et al., 2001). Favoured feeding grounds for lapwing and redshank are small scaled drainage features such as drains and rills crossing wetlands such as wet grassland (Eglington et al., 2008; Milsom et al., 2002). Softer ground allows snipe to forage for food easily (Ausden et al., 2001; Smart et al., 2008). The positive correlation between wetness and the probability of nesting species of waders has also been shown by Milsom et al. (2000) (Figure 1-15).

Waterlogged features support a higher biomass of surface-active and aerial invertebrates, although species numbers decrease with flood events (Plum, 2005). Wet

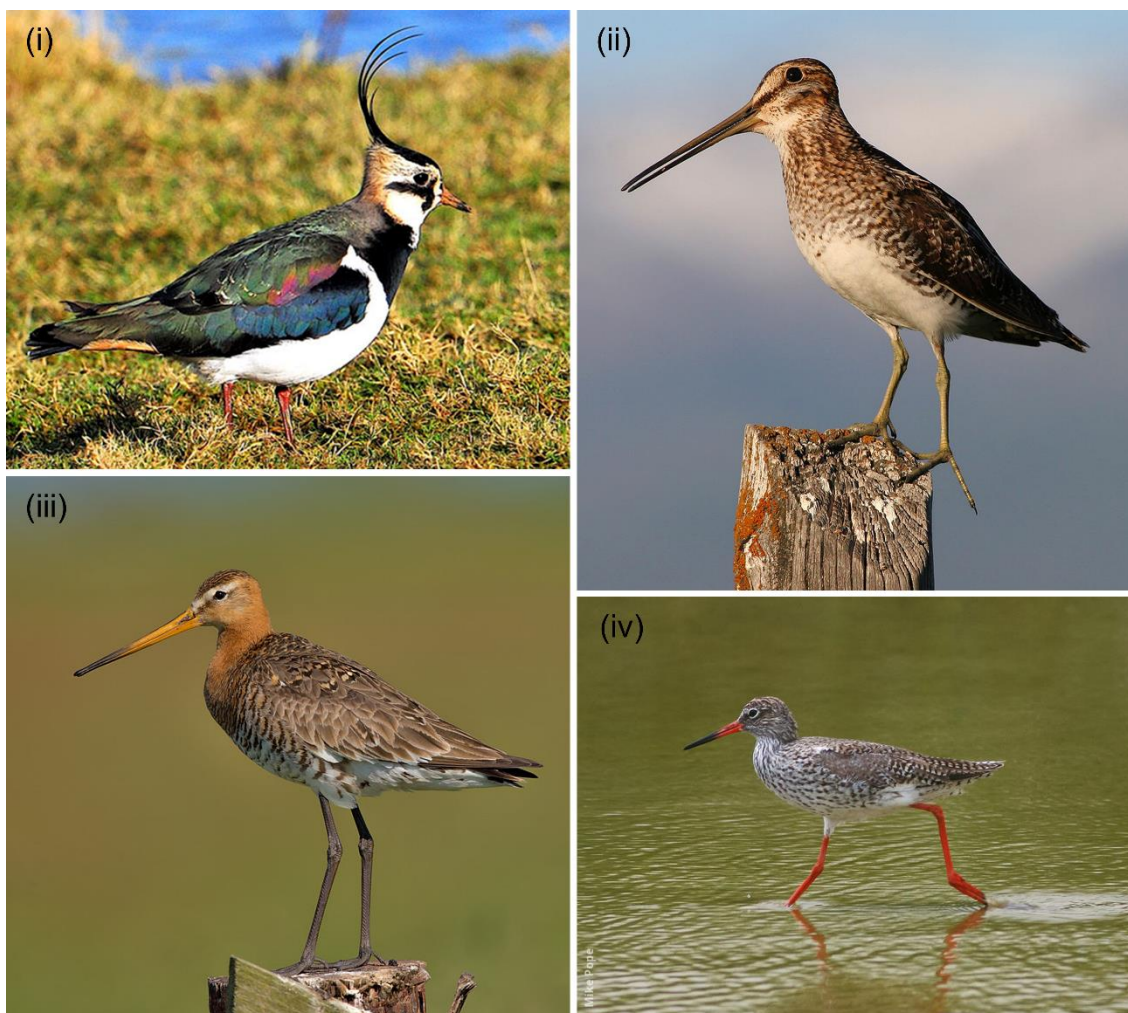


Figure 1-14. Wading birds in UK decline: (i) lapwing (*Vanellus vanellus*) © John Sheppard, (ii) snipe (*Gallinago gallinago*) © Sean Brezeal, (iii) black-tailed godwit (*Limosa limosa*) © Mike Pope, and (iv) redshank (*Tringa totanus*) © Andreas Trepte

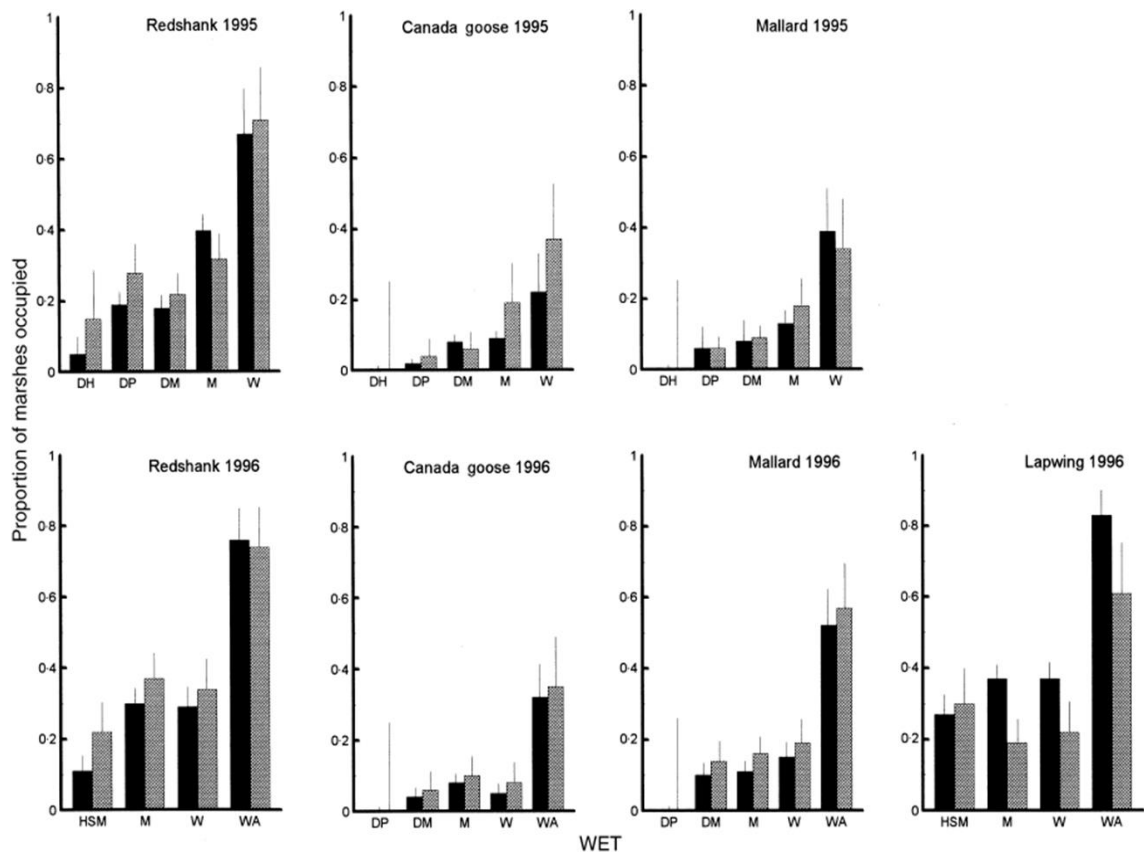


Figure 1-15. Relationships between observed (solid bars) and predicted (cross-hatched bars) proportion of marshes occupied and wetness of rills in early June (WET) for redshank, lapwing, Canada goose and mallard. DH: dry/hard, DP: dry/penetrable, DM: dry/moist, M: moist, W: wet, WA: some water and water categories pooled. In the lapwing and redshank models, the three driest categories were pooled due to small sample sizes to produce a HSM category (Milsom et al., 2000)

footdrains and wet pools in eastern England were found to support a greater abundance of aerial invertebrates than surrounding grazing marshes (Eglington et al., 2010) (Figure 1-16). Also in the UK, the distribution of the near threatened Desmoulin's whorl snail (*Vertigo moulinsiana*) (Killeen et al., 2012) has been directly linked to water levels (Tattersfield and McInnes, 2003). Temporary ponds support fewer macroinvertebrates than permanent ponds, although with a higher proportion of rare species (Collinson et al., 1995). Community structure also differs due to adaptive abilities of species to cope with drying phases (Tarr et al., 2005), with the frequency of such drying events being significant (Whiles and Goldowitz, 2001). Macroinvertebrates that successfully exploit ephemeral ponds will, most likely, be able to tolerate a range of changes in water levels.

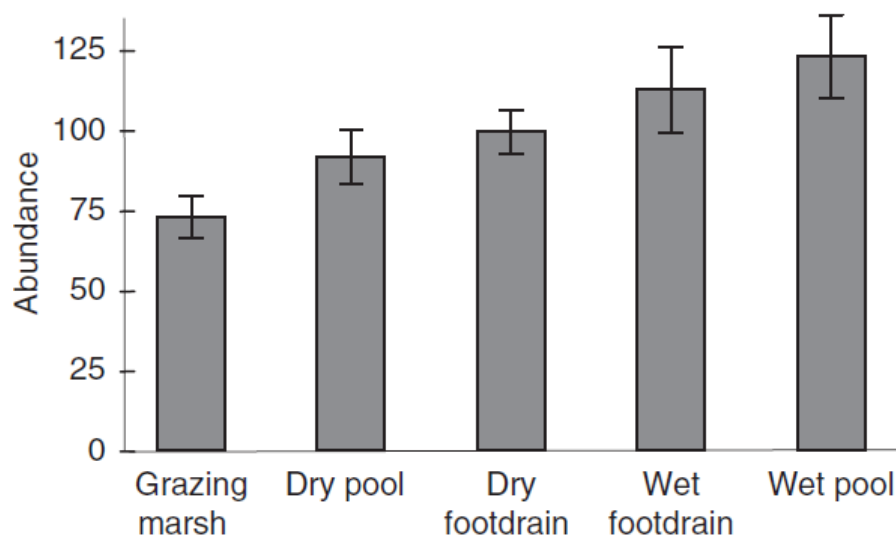


Figure 1-16. Abundance of aerial invertebrates from pitfall traps and sticky traps across five wet grassland habitats (after Eglinton et al., 2010)

1.3.2.2 Groundwater and wetland vegetation

Groundwater flux can exert strong controls upon the hydrological regime (Section 1.3.1), nutrient status, and species composition (Wheeler et al., 2009). Water chemistry is a principal factor in determining the ecological composition of different wetland habitats (Wassen et al., 1989). Water flowing through Chalk or limestone aquifers dissolves minerals such as calcium, sodium, bicarbonate and chloride (Acreman and Miller, 2006), becoming base-rich. Hence, chemical properties of groundwater are largely distinct from surface water. The spatial distribution of nutrients and other chemical agents produces and maintains conditions for specific plant species and communities (Klijn and Witte, 1999). Groundwater-fed wetlands are characterised by different floral communities to those fed by surface water or precipitation alone.

Common vegetation types associated with lowland floodplain wetlands in the UK include common reed (*Phragmites australis*), tall sedges (*Carex* spp.), purple loosestrife (*Lythrum salicaria*) and yellow iris (*Iris pseudacorus*) (McBride et al., 2011) (Figure 1-17). Chalk or limestone bedrock provides low nutrient calcareous groundwater, characterised by shorter vegetation and a diversity of sedges and mosses. Lowland valley wetlands in such areas may have an abundance of black bog-rush (*Schoenus nigricans*) tussocks, although in the southeast of the UK examples are restricted to highly calcareous situations. Wetlands with multiple water sources, transitional between alkaline and acidic conditions, can be highly diverse, yet are typically sedge dominated. Other characteristic plant species include bog bean (*Menyanthes trifoliata*), water horsetail (*Equisetum*

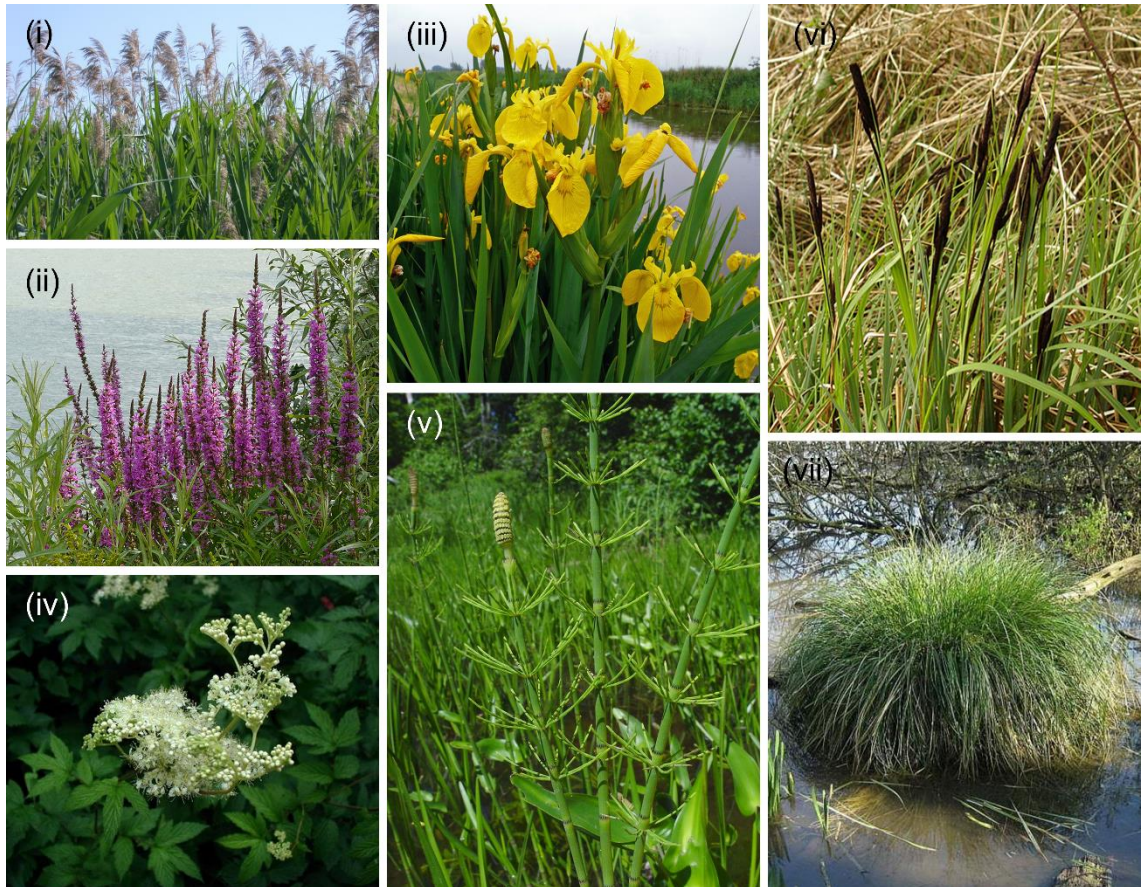


Figure 1-17. Characteristic species of lowland floodplain wetlands: (i) common reed (*Phragmites australis*) © Wasyl Bakowsky, (ii) purple loosestrife (*Lythrum salicaria*) © Manfred Heyde, (iii) yellow iris (*Iris pseudacorus*) © LilliesWaterGarden; lowland floodplain wetlands with groundwater and surface water inputs: (iv) water horsetail (*Equisetum fluviatile*) © Donald Cameron, (v) meadowsweet (*Filipendula ulmaria*) © Sten Porse; and groundwater fed standing water: (vi) lesser pond sedge (*Carex acutiformis*) © John Somerville, (vii) and tussock sedge (*Carex paniculata*)

fluviatile), marsh cinquefoil (*Potentilla palustris*), meadowsweet (*Filipendula ulmaria*), devil's-bit scabious (*Succisa pratensis*), marsh bedstraw (*Galium palustre*) and common valerian (*Valeriana officinalis*). Areas of permanent standing water in wetlands fed by calcareous groundwater are often associated with swamps dominated by single species, such as greater and lesser pond sedges (*Carex riparia* and *Carex acutiformis*), tussock sedge (*Carex paniculata*) or common reed.

Even a small supply of groundwater can cause significant differences in the nutrient budget and chemical environment of wetlands (Acreman and Miller, 2006), which in turn will result in ecological adjustments. For example, changes to the ecological composition of Wicken Fen (Cambridgeshire, UK) were initially attributed to drying out by drainage and groundwater abstraction (McCartney and De la Hera, 2004). Yet, it was later shown

that reduced inundation due to flood management from the base-rich groundwater fed river passing through the site had altered the acidity of the wetland.

1.3.3 Investigating groundwater/surface water interaction in wetlands

Investigative techniques are required that can capture processes of groundwater/surface water interaction at a high spatial resolution. Such techniques should be non-destructive and minimally invasive due to the sensitivity of many wetland environments. Furthermore, resource restrictions often compel economical approaches. Temperature is a useful natural groundwater tracer which can identify small-scale zones of groundwater discharge (Anderson, 2005; Conant, 2004). Heat is propagated in soils by conduction through the pore water and solid matrix, and advection by pore water flux. In isotropic, homogenous, and saturated soils, heat transfer is affected by the specific heat and density of the fluid and fluid/solid system, fluid velocity, and thermal conductivity (Bredehoeft and Papaopulos, 1965). Relatively stable groundwater temperatures are transported by advection of upwelling groundwater, enabling its distinction from fluctuating surface water temperatures. A significant body of research is centred on identifying and quantifying localised groundwater flux in the hyporheic zone (Briggs et al., 2012; Constantz et al., 1994; Hannah et al., 2009; Keery et al., 2007; Krause et al., 2012; Schmidt et al., 2007). However, applications to wetlands are relatively rare, and include incorporation of a small number of vertical profiles into 1D and 3D flow and heat transport models (Bravo et al., 2002; Hunt et al., 1996); use of single point measurements at different depths by location to constrain 2D simulations (Burow et al., 2005; Gamble et al., 2003); inference of the control of advection and conduction on vertical heat transfer within a single profile (McKenzie et al., 2007); and, qualitative assessment of temporal fluctuations in groundwater flux by comparison of a single point measurement of soil temperature to surface water and air temperature (Zapata-Rios and Price, 2012). Small-scale variability in temperature has been considered along the horizontal plane of a cross-fault ditch (Bense and Kooi, 2004). Nevertheless, temperature observations in such studies are frequently restricted to a limited number of point measurements (Bravo et al., 2002; Hunt et al., 1996).

Water chemistry and isotopes can be supplemental in delineating areas of groundwater discharge (Kehew et al., 1998). Groundwater is, for the most part, chemically distinct from surface water (Wassen et al., 1990) (see Section 1.3.2.2). Differences in groundwater flow paths, geochemical reactions and residence times will correspond to differences in hydrogeochemical composition (Brunke and Gonser, 1997; Soulsby et al., 2007). Reactants and their reactive products may be used to identify distinct

geochemical zones, as shown by Gooddy et al. (2002), who identified zones of reduction distinct from denitrification below two lagoons in the Chalk aquifer of southern England. The chemical signature of water also allows potential definition of different sources (Kendall and McDonnell, 2012). Geochemical tracers have been used to assess water sources in geologically complex catchments. For example, alkalinity was used to show a highly heterogeneous system of within-catchment groundwater and surface water flow pathways in the catchments of the upper River Severn (Neal et al., 1997). Soulsby et al. (2007) also used alkalinity along with chloride to infer the interactions between groundwater and surface water in the Girnock Burn catchment, Scotland, where local variability implied marked differences in groundwater flow paths, residence times and geochemical reactions.

Botanical indicators of groundwater also have the potential to provide a cost-effective means of site characterisation (Lewis, 2012). However, links between phreatophytes (plants which draw water from the water table) and groundwater discharge are more established in arid and semi-arid regions than more humid regions (Batelaan et al., 2003). Plants have, however, been regarded as indicators of groundwater discharge within wetlands in the Netherlands (Grootjans et al., 1988; Klijn and Witte, 1999; Lucassen et al., 2006; Schot et al., 1988; van Diggelen et al., 1988; Wassen et al., 1988; Wierda et al., 1997) and Minnesota, USA (Almendinger and Leete, 1998; Batelaan et al., 2003; Glaser et al., 1990; Goslee et al., 1997; Rosenberry et al., 2000). One species especially, marsh marigold (*Caltha palustris*) (Figure 1-18), has been reported to be a reliable indicator of groundwater discharge across temperate regions (Klijn and Witte, 1999; Lewis, 2012; Rosenberry et al., 2000; Wierda et al., 1997). However, the use of a particular species as an indicator of groundwater is area-restricted, and dependent on local geochemistry and soil characteristics. Research in the UK has focussed on general water requirements of wetland plant communities and species (Gowing et al., 2002; Newbold and Mountford, 1997; Wheeler et al., 2009) (see Section 1.3.2.1). Specifically, botanical indicators for groundwater are not defined in the UK.

1.3.4 Wetland management

Current and historical wetland management practices revolve around manipulating water levels and flow to meet regulatory requirements and/or maintain processes and services, such as the conservation of desired species or communities, flood mitigation, water quality and supply, and arable or pastoral productivity (Morris et al., 2008). Vegetation and nutrient management are also crucial, although the exact strategy depends on the



Figure 1-18. Marsh marigold (Caltha palustris) © Arthur Haines, which has been used as an indicator of groundwater discharge in temperate regions (Lewis, 2012)

resources available and final goal (McBride et al., 2011). The success of any management strategy rests on the impact to the wetland ecosystem.

In the UK, attempts to tap into the pool of knowledge on wetland restoration and ecohydrology have resulted in frameworks for the maintenance and restoration management of wetlands (Mountford et al., 2005; Wheeler et al., 2004; Wheeler et al., 2009). These have built upon specific agri-environment schemes and site protection designations (Acreman and Mountford, 2009), for instance in grazing marsh (Mountford, 1994) and floodplain grasslands (Gowing et al., 2002). The guidelines provide information on the hydrological regime favourable for a range of habitats through monthly maximum and minimum water depths, plus information on suitable depth durations.

Management strategies often focus on the condition of the wetland to achieve the required objectives. These may be a continuation or modification of an existing strategy or the implementation of a new strategy or restoration programme (McBride et al., 2011). A range of specific options and tools are available to the wetland manager to achieve the required outcome, some of which are listed in Table 1-7. Water-level manipulation is generally achieved through use of control structures such as ditches, weirs, sluices, aqueducts and pumps. Other techniques for wildlife conservation may be categorised into natural management, such as the use of seed banks, plant succession and herbivory, and artificial management, including planting, ditching, scrape creation and island building (Baldassarre et al., 2006). Agricultural management of wetlands, aside

Table 1-7. Wetland management strategies and options (after Benstead et al., 1999)

Wetland condition	Management strategy	Management options	
		Water management	Vegetation management
Good	Maintain existing strategy	Determine hydrological regime	Identify existing regime
		Maintain current hydrological regime	Maintain current vegetation management
Moderate	Improve/modify strategy	Monitor water levels and quality	Monitor vegetation change
		Investigate possibility/desirability of return to natural regime	Manage scrub and trees
Poor	Implement strategy or creation/restoration programme	Block or remove drains	Cut back vegetation to remove standing crop and litter
		Possibility of lowering wetland surface	Controlled burn
		Prevent inflow of nutrient-rich water. Options including silt trap, creation of nutrient trapping wetland upstream, sympathetic management of surrounding land	Introduce cutting regime
		Monitor water levels and quality	Introduce grazing animals
		Identify water source, chemistry and nutrient status	Monitor vegetation change
		Reinstate natural regime	
		Engineering options including bunding, sluices, ditch creation, earthworks	Assess feasibility of natural regeneration
		Monitor water levels and quality	Soil remediation
			Planting seeds from nearby sources
			Introduce cutting regime
			Introduce grazing animals
			Monitor vegetation development

from drainage, may include control of natural flood flows for rice, shellfish and fish crops, or induced flooding to encourage hay crop growth (Everard, 2005). Practises range from management of whole systems to intervention for specific species or features.

Redgrave and Lopham Fens (Figure 1-19), one of the largest of England's valley wetlands which is located between Norfolk and Suffolk, provides an example of the



Figure 1-19. Map of Redgrave and Lopham Fen, with inset showing scrape removal of peat © Helen Smithy

implementation of multiple management techniques (Seccombe, 1993). The site is notable as providing habitat for one of only three populations of the fen raft spider (*Dolomedes plantarius*). Before the 1950s the water regime was fed heterogeneously by groundwater and surface water. A complex geology produced nutrient poor waters with highly varied pH and base-content, which was reflected in different types of fen vegetation. Management in the 1950s and 1960s included deepening of the River Waveney that runs through the wetland and groundwater abstraction for public supply. These led to a change in the water regime to a dependence on winter storage of precipitation and flooding. Loss of plant communities and scrub invasion were aggravated by nutrient release from drying peat. Restoration work through the 1990s and 2000s involved relocation of the abstraction borehole, construction of a sluice and flood embankments in the River Waveney, scraping of rotted and enriched peat, scrub removal, planting, vegetation cutting and grazing. Outcomes have included groundwater recovery, exceptional botanical species recovery, water quality improvements and survival of the fen raft spider.

In the carboniferous limestone region of central Anglesey, the Anglesey Fens (Figure 1-20) contain the most extensive region of alkaline fen in western Great Britain (Natural Resources Wales, 2015). Past peat cutting and drainage resulted in extensive loss of wetland area. Important actions for restoration works included mowing, harvesting and



Figure 1-20. View across Anglesey Fens and scrub clearance as part of the restoration works © Natural Resources Wales

introducing a conservation grazing scheme combined with scrub clearance. Nutrient inputs were reduced through agreements with farm and landowners, the blocking of drains raised water levels and established pathways between groundwater source areas and the fen surface in combination with ditch re-establishment, whilst peat stripping was undertaken to restart the succession processes. These measures are currently ongoing, yet illustrate the large range and complexity of techniques that may be necessary for wetland management.

1.3.5 Modelling wetland hydrology

The impacts of abstraction, sustained low river flows, climate change, or feedback from water management activities taking place within the catchment could result in significant adverse impacts upon wetland hydrology and, in turn, ecological conditions. Where

underlain by permeable geology, such as chalk, wetlands may be particularly vulnerable to such changes, as illustrated by examples such as Redgrave and Lopham Fens and others in Section 1.3.4. An ability to accurately predict the impacts of environmental changes, whether they are climate induced, or result from groundwater abstraction, flood management, or some other process, is vital for wetland management where species conservation and ecosystem service provision relies on managing hydrological functions (Acreman et al., 2009). Due to the complexity of process interactions in these wetlands, quantifying a water balance through field observations alone is often impractical (see Section 1.3.1.1). Models able to accurately represent wetland hydrology will enable the assessment of possible degradation to wetland ecosystems through environmental change (Acreman and Jose, 2000; Thompson et al., 2004). In turn, such models will permit assessment of the likely success of modifications to wetland management designed to mitigate the impacts of climate change.

Models explicitly for wetlands have arisen around specific sites and motives. To illustrate, for a cypress pine forest wetland in Florida, Mansell et al. (2000) used WETLANDS, a multidimensional water flow and solute transport model which provides a dynamic link between pond water, groundwater, and unsaturated soil zones. Developed for such regional forest wetlands, the model does not, however, incorporate overland and channel flow, or regional groundwater flow. Su et al. (2000) adapted SLURP, a semi-distributed model originally developed for simulating streamflow (Kite and Singh, 1995), to predict water levels of a 3 ha prairie wetland in Saskatchewan, Canada with good accuracy, but encountered poor translation to a 2400 m² smaller scale wetland (Figure 1-21). Fluctuations in the water levels during the entire period were simulated poorly, with a low R² value (0.37) as indicated on the scatter plot. However, the effect of wet and dry years on water levels was reasonable.

Kazezyilmaz-Alhan and Medina (2008) simulated the hydrology of a restored wetland in Sandy Creek, North Carolina, using WETSAND. The model is semi-distributed and incorporates surface groundwater interactions, surface flow, solute transport, and is able to account for upstream urban contributions. However, it is difficult to assess the performance of the model as no calibration or validation details were presented.

In a comparative study, two semi-distributed models, SLURP and WATFLOOD, were used to assess the feasibility of modelling the Hudson Bay Lowlands subarctic wetland, Canada (Jing et al., 2010). Differences in model structure, hydrological process representation, parameters and simulation performance were highlighted, along with

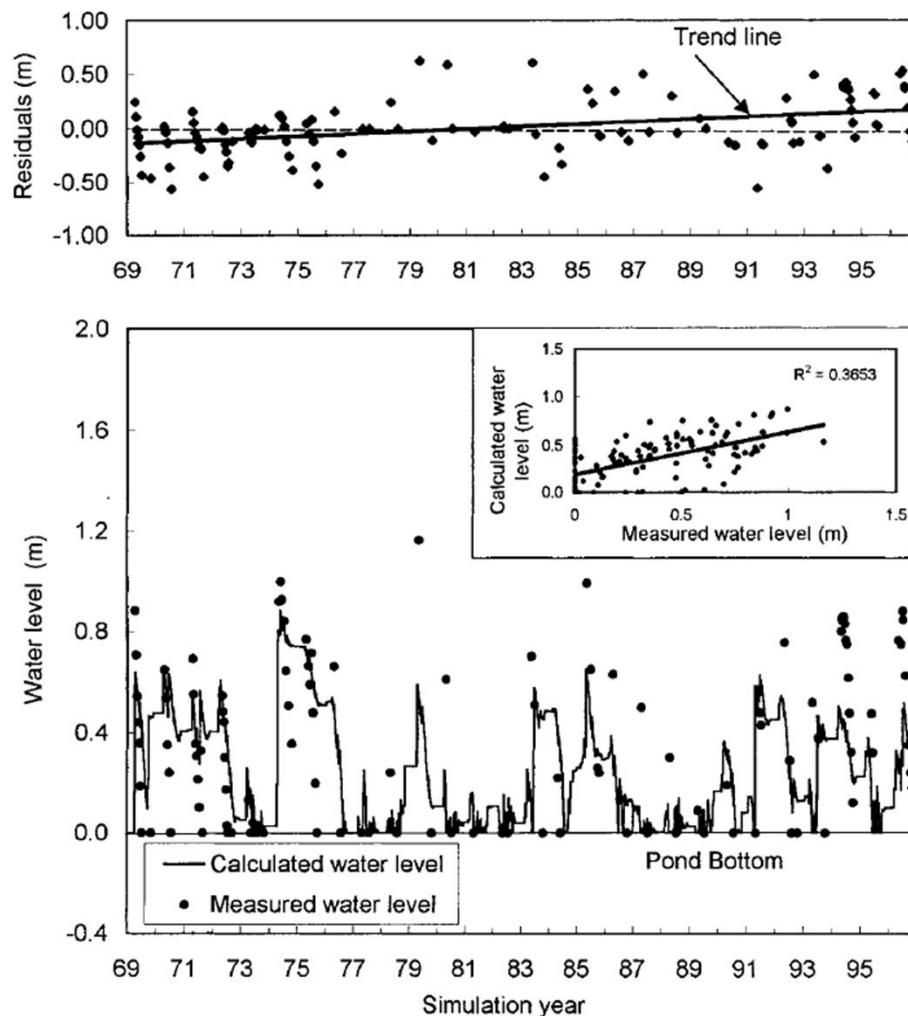


Figure 1-21. Simulated water levels of a small scale (2400 m²) wetland in Saskatchewan, Canada using a modified version of the semi-distributed SLURP (Su et al., 2000)

modifications needed to improve accuracy (Figure 1-22). Snowmelt and peaks of spring runoff simulated by SLURP were earlier than those simulated by WATFLOOD, due to the exponentially increasing snowmelt rate adopted by SLURP. Both models underestimated the peaks of spring runoff. SLURP slightly underestimated summertime evapotranspiration, whilst this flux was overestimated by WATFLOOD.

A more comprehensive understanding of wetland hydrological processes requires full representation of spatial and temporal variations in wetland water table gradients. As wetland soils can be regarded as shallow aquifers (Acreman and Miller, 2006), groundwater flow models have been used for such a purpose. Most groundwater flow models use general head boundary nodes to represent wetlands (Merritt, 1997), or otherwise employ constant head nodes, stream routing, and lake stage packages

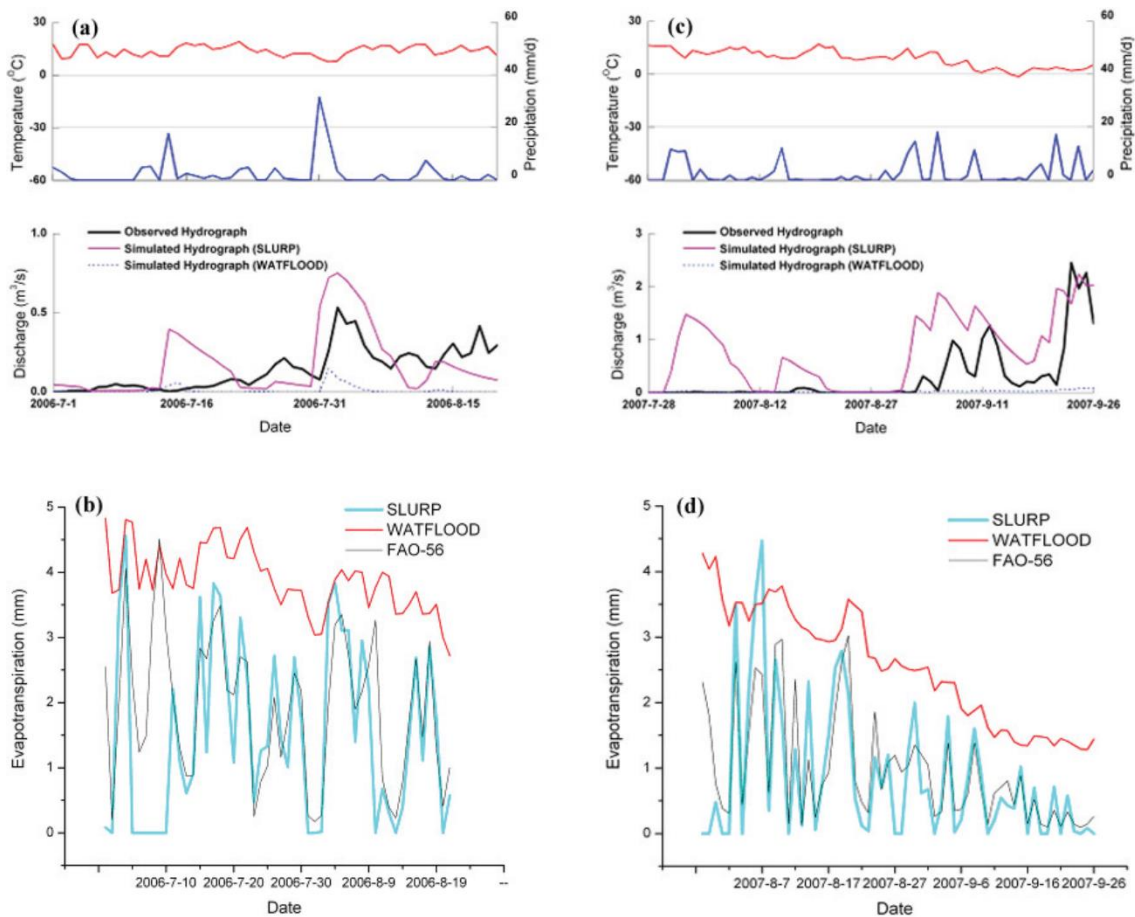


Figure 1-22. Comparisons between SLURP and WATFLOOD at the Hudson Bay Lowlands, Canada: (a) Simulated and observed daily hydrographs, (b) daily evapotranspiration in 2006; and, (c) simulated and observed daily hydrographs, (d) daily evapotranspiration in 2007 (Jing et al., 2010)

(Restrepo et al., 1998). The three-dimensional, finite difference USGS groundwater flow model, MODFLOW, has been relatively well implemented in this context. For example, as detailed in Table 1-8, Gerla and Matheney (1996) employed MODFLOW to simulate vertical saturated flow for a 500 ha wetland in the Red River Valley of eastern North Dakota, USA. Reeve et al. (2000) used data from the Hudson Bay Lowland and Glacial Lake Agassiz peatlands to highlight the influence of hydrogeological setting and role of substrate permeability in controlling vertical hydraulic gradients. Bradley (2002) simulated successive hydroperiods to describe water table fluctuations in a floodplain wetland, Narborough Bog, UK and their relationship to river stage. Bradford and Acreman (2003) applied MODFLOW to ascertain the dominant influence of precipitation and evaporation over ditch stage for a single field in the wet coastal grasslands of the Pevensy Levels, UK. Whiteman et al. (2004) modelled the effect of abstraction on

Table 1-8. Summary of relevant studies using MODFLOW to model wetland hydrology

Source	Wetland, size and location	Driving data	Resolution / grid size	Key results
Gerla and Matheney (1996)	Lunby and Stewart wetland, North Dakota, 500 ha	Groundwater head, precipitation, evapotranspiration	1D vertical section, 1 m ² base area vertical columns, vertical discretisation 1 m	Good correspondence between observed and simulated hydrographs. Shallow vertical groundwater flow dominates the hydrological budget
Reeve et al. (2000)	Hudson Bay Lowland, Canada, 10000 m profiles	Groundwater head and recharge, evapotranspiration	2D profile, 100 m to 130 m cells	Vertical ground-water flow primarily controlled by mineral soil permeability. When peat forms over low permeability mineral soil, vertical movement becomes negligible and lateral flow of water the peatland hydrology
Bradley (2002)	Narborough Bog, UK, 9.5 ha	Groundwater head, precipitation, evapotranspiration, river stage	10 m x 10 m	At low flows, the water table grades to the river levels, which acts as a head control. At high flows water levels are elevated due to the presence of a silt-clay levee
Bradford and Acreman (2003)	Pevensey Levels, UK, 75 km ²	Groundwater head, precipitation, evapotranspiration, channel stage	400 m x 400 m	Data insufficient for full calibration. Evaporation and rainfall dominate water balance.
Whiteman et al. (2004)	Great Cressingham Fen, UK, 13.7 ha	Groundwater head, precipitation, evapotranspiration, channel stage	200 m x 200 m	Difficulty in representing shallow water tables for estimation of ecological effects. Typical cell size of the model results in a prediction of groundwater head at greater depth than rootzone.

drawdown in Great Cressingham Fen in support of an ecohydrological impact assessment.

A shortcoming of such groundwater models is that surface water, vegetation characteristics and evapotranspiration are often not incorporated. Restrepo et al. (1998) developed a wetland module to incorporate within MODFLOW. The main development

of the wetland simulation model is the representation of surface flow through wetland vegetation. Sheet flow and channel flow are solved by flow routing using a modified Manning equation. However, the module was developed for swamp areas rather than wetlands with diverse vegetation.

Finite element approaches, such as used in FEFLOW (Diersch, 1993) and HPP-GMS (Sanchez-Perez and Tremolieres, 1997), provide a variant to finite difference spatial discretisation, as used in MODFLOW. The finite difference method computes values for nodal points set either at the intersections of a rectangular grid mesh or at the centre of each grid cell or block (Jones, 1997). The finite element method calculates values at nodes as well as interpolating values between nodes. Garoute et al. (2009) used FEFLOW to assess the potential impact of a planned canal on the estuary wetlands of the River Seine, France, and to design mitigating measures. Although the finite element approach increases model sophistication, the increase in complexity can cause problems of extreme data demand and inability to calibrate outputs, poor translation of physical equations to large spatial scales, and unrealistic computational demands (Beven and Binley, 1992).

Wetland hydrology is often the product of complex surface-groundwater interactions, and necessitates models that incorporate the two. This may be achieved through use of an integrated surface groundwater model, such as that described by Crowe et al. (2004), who developed a 2D simulation of groundwater-wetland interaction and contaminant transport for Point Pelee marsh and Lake Erie, Canada. The model performed well, confirming that reversals in the direction of groundwater flow were due to seasonal fluctuations in marsh and lake water levels (Figure 1-23). Barr and Barron (2009) provide another example in the use of MODHMS to identify the contributions of surface water and groundwater to runoff and wetlands in the Southern River catchment, Australia. Major processes governing hydrological and hydrogeological conditions were found to be dependent on soil type, topography and the seasonal distribution of rainfall. Frei et al. (2010) used the fully integrated surface-subsurface model HGS to examine the effects of micro-topographical hollows and hummocks on surface groundwater exchange and runoff production for a floodplain wetland in Lehstenbach, Germany. The micro-topography buffered rainfall inputs, producing a hydrograph predominantly characterised by subsurface flow. However, shifts to surface flow dominating occurred during intense rainstorms (Figure 1-24).

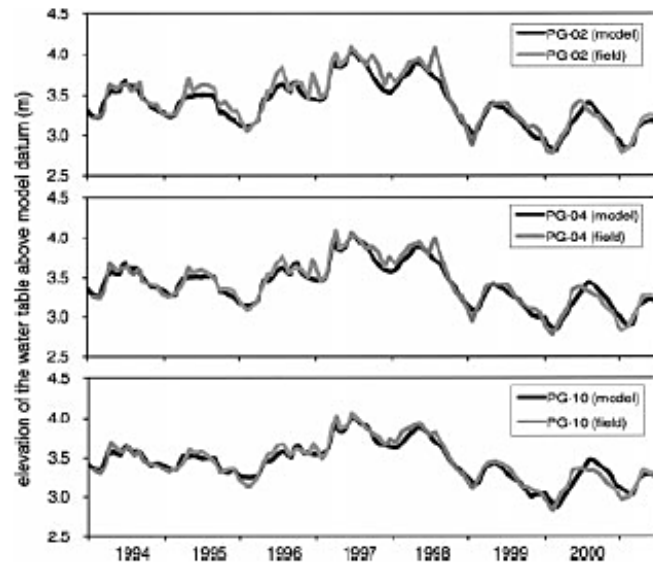


Figure 1-23. Observed and simulated water table elevations for selected dipwells located close to Lake Erie (PG-02), the Pont Pelee marsh (PG-10), and intermediate between the two (PG-04) (Crowe et al., 2004)



Figure 1-24. A floodplain wetland in Lehstenbach, Germany experiencing surface flow during a storm event in 2009 (Frei et al., 2010)

Separate surface water and groundwater models may be coupled together. Krause and Bronstert (2004) used a coupling of the surface and soil water model WASIM-ETH-I and groundwater model MODFLOW to quantify surface groundwater fluxes in a lowland subcatchment of the Havel, Germany. Such coupled hydraulic and groundwater models simulate the wetland as part of a much larger hydrological system (Acreman and Miller, 2006).

A prominent and widely used model coupling for wetland hydrology is that of the hydraulic model MIKE 11 with the hydrological model MIKE SHE. Refsgaard et al. (1998) established the coupling to assess the impacts of a hydropower scheme on Danubian Lowland wetlands in Slovakia and Hungary. Explicit factors of flood frequency and duration, depth of flooding, depth to groundwater table, capillary rise, flow velocities, sedimentation and river water quality were quantified, enabling the evaluation of different water management schemes. The approach was likewise applied by Thompson et al. (2004) to the Elmley Marshes, southeast England. The same model was used to simulate impacts of water level management (Thompson et al., 2004) and to evaluate the effect of climate change on these marshes (Thompson et al., 2009). Staes et al. (2009) employed MIKE SHE/MIKE 11 to study various methods for wetland habitat restoration, and to assess potential hydrological impacts of each approach. Scenarios included infiltration restoration, restoration of downstream valley bottom wetlands, restoration of upstream valley bottom wetlands, and restoration of upstream headwater wetlands. River valley rewetting was found to decrease stream flow, and increase saturated zone flow, promoting groundwater recharge.

Comprehensive, contemporary studies on individual wetlands have relied on simulation of hydrological processes within fully integrated or coupled groundwater/surface-water models (Crowe et al., 2004; Frei et al., 2010; Krause and Bronstert, 2005; Refsgaard et al., 1998; Thompson et al., 2009; Thompson et al., 2004). However, these modelling studies often contain simple interpretations of the saturated zone through single layer lithology (Frei et al., 2010; Refsgaard et al., 1998; Thompson et al., 2009; Thompson et al., 2004) or transfer functions (Krause and Bronstert, 2005). Where applied to wetlands with more complex subsurface hydrogeological structures, processes have been partially represented as boundary conditions (Crowe et al., 2004). To the author's knowledge, there are no wetland applications of fully distributed groundwater/surface-water models to compound geologies.

1.4 Climate change and wetlands

1.4.1 Impacts of climate change on wetland hydrology

Unequivocal warming of the climate (IPCC, 2014) will impact the global hydrological cycle (Arnell and Gosling, 2013), with implications for aquatic ecosystems (Matthews and Quesne, 2009; Poff et al., 2002) and water resources (Gosling et al., 2011; Oki and Kanae, 2006). Wetlands are highly vulnerable to climate change due to the primary importance of the hydrological regime in controlling their ecological characteristics (e.g. Baker et al., 2009) (Section 1.3.2). Climate change will alter precipitation and evapotranspiration rates which, in turn, will result in changes to runoff and groundwater levels. The key roles of these processes in controlling wetland vegetation (Baldwin et al., 2001; Wheeler et al., 2009), animals (Ausden et al., 2001; McMenamin et al., 2008) and biogeochemical cycling (Lischeid et al., 2007; McClain et al., 2003) means that climate change is likely to have major impacts on the world's wetlands, their flora and fauna as well as delivery of the many ecosystem services which they provide. In addition, the importance of flow regime in controlling processes of water quality, sediment transport, dissolved oxygen levels, plus type and distribution of habitat (Bunn and Arthington, 2002; Poff et al., 1997; Richter et al., 1998; Warren et al., 2015) means that climate change is likely to have a major impact on fluvial ecosystems, their biota, and the many services which they provide.

To illustrate, probabilistic projections from UKCP09, which provides climate change projections for the UK, show, under a medium emissions scenario for the 2080s, increases in winter precipitation for central probability levels (as likely as not to be exceeded) are in the range +10 to +30% over the majority of the country (Murphy et al., 2009) (Figure 1-25). Summer decreases show a south to north gradient, from almost 40% in southwest England to almost no change in the northeast. However, there is a great deal of uncertainty. Changes at 10%, 50% and 90% probability levels have different magnitudes and may be in different directions, becoming wetter or drier. For example, although summer precipitation is projected to decrease almost everywhere in the UK at the 10% and 50% probability levels, at the 90% probability level they are projected to increase.

National changes in UK river flows and regional chalk groundwater levels have been estimated from projections using an ensemble of 11 variants of the Met Office Regional Climate Model (HadRM3), itself used to drive UKCP09 probabilistic projections, under

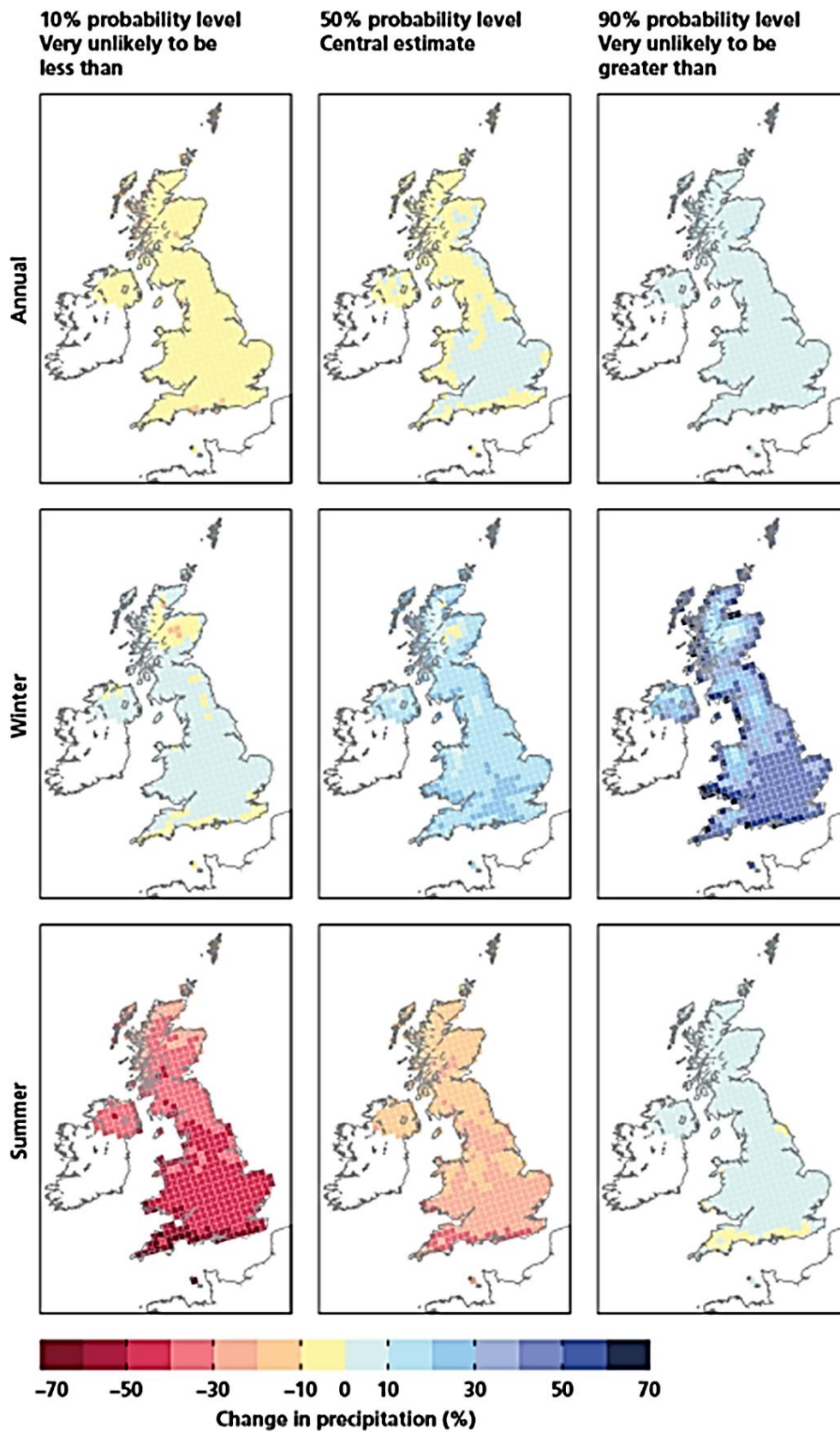


Figure 1-25. Changes (%) in annual (top), winter (middle) and summer (bottom) mean precipitation at the 10, 50 and 90% probability levels, for the 2080s under the Medium emissions scenario (Murphy et al., 2009)

the Future Flows and Groundwater Levels project (Jackson et al., 2011; Prudhomme et al., 2012). Changes are only associated with a medium emission scenario, although some climate variability and uncertainty is captured by the ensemble, with each variant equally likely. For river flows, time series representative of the future climate were used to drive the hydrological model CERF. Changes in mean annual flow for the 2050s are generally within 20% of each other (Figure 1-26). Small increases, mainly in the south and east, are shown for roughly half of the scenarios. Though, one scenario suggests decreases of up to -40% for the south and east. Under the majority of the scenarios, the west is projected to see a reduction in mean annual flow up to -40%, while in Scotland there is little change.

Hydrological impacts of climate change are commonly evaluated by using climatic projections, derived from forcing General Circulation Models (GCMs) with alternative emissions scenarios, to drive hydrological models. Examples of this approach cover hydrological systems at various scales from global assessments (Arnell, 2003; Arnell and Gosling, 2013; Gosling et al., 2010; Nohara et al., 2006), through regional (Arnell, 1999) and national scales (Andréasson et al., 2004), major river basins (Conway, 1996; Nijssen et al., 2001; Thompson et al., 2013), medium and small catchments (Chun et al., 2009; Thompson, 2012), down to individual wetlands within catchments (Thompson et al., 2009).

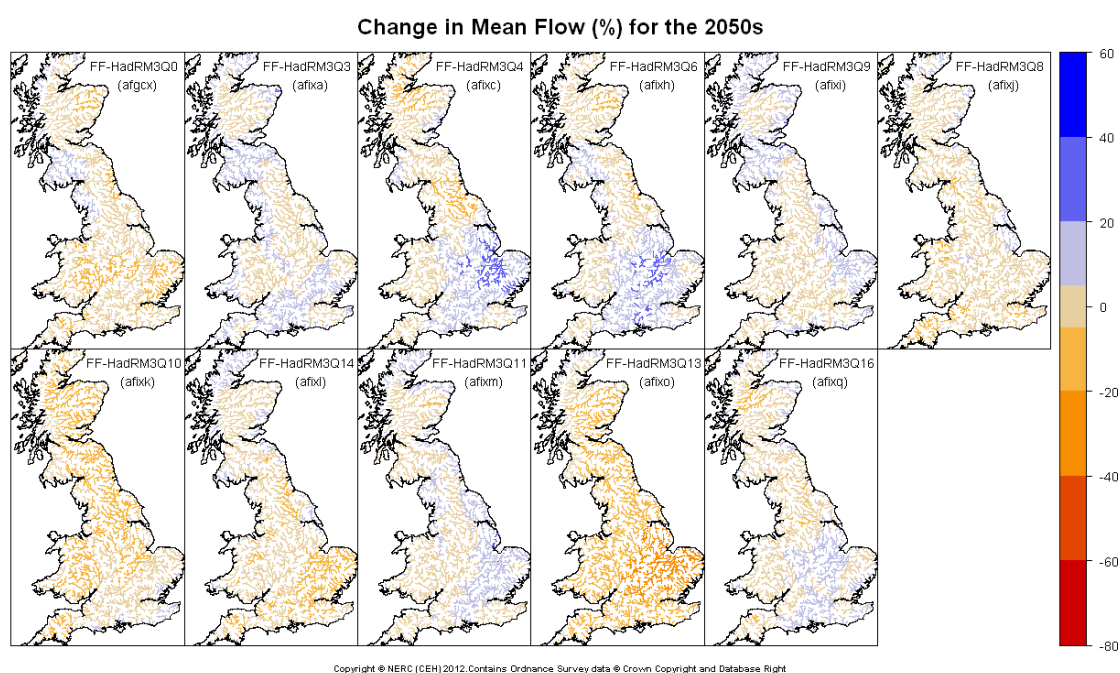


Figure 1-26. Changes in mean annual flow for the 2050s under 11 scenarios from the Future Flows and Groundwater Levels Project (Prudhomme et al., 2012)

Changes in groundwater levels for the Chalk aquifer of the Marlborough and Berkshire Downs and south-west Chilterns were simulated using a British Geological Survey (BGS) ZOOMQ3D regional groundwater model (Haxton et al., 2012; Jackson et al., 2011). For selected observation boreholes across the aquifer, reductions in groundwater level were simulated under all but three scenarios (Figure 1-27). Scenario averages (squares in Figure 1-27) range between a maximum decline -2.7 m at Stonor Park to no change at Great Park Farm.

Wetlands that interact with other water bodies will be impacted by climate change-driven hydrological changes occurring over potentially much larger catchment areas (Baker et al., 2009). The response of wetlands to such changes is a complex balance between adjustments in water levels, temperature, nutrient cycling, physiological acclimation and

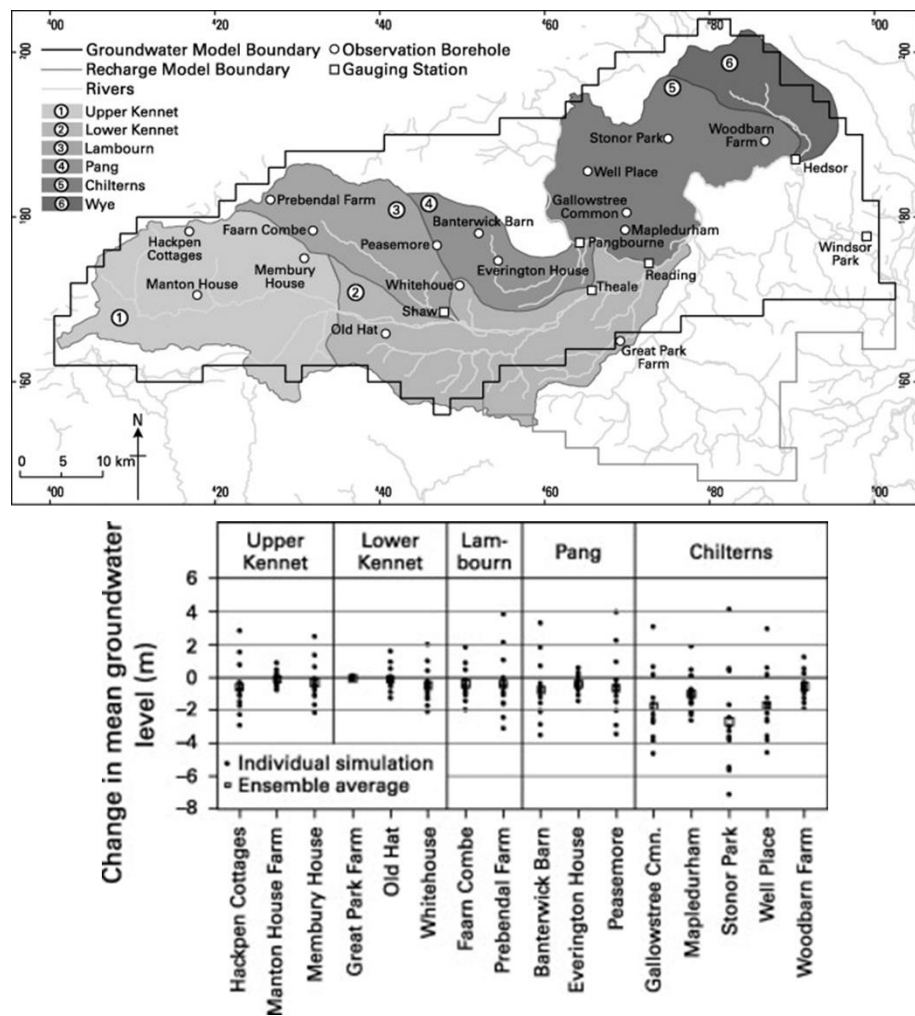


Figure 1-27. Observation borehole locations and changes in mean groundwater level for the 2080s for the Chalk aquifer of the Marlborough and Berkshire Downs and south-west Chilterns simulated under 11 scenarios from the Future Flows and Groundwater Levels Project (Jackson et al., 2011)

community reorganisation (Oechel et al., 2000). The effects of climate change on regional aquifers and catchment runoff may cause intricate and significantly detrimental impacts to wetlands underlain by permeable geology, such as the chalk lowlands of southeast UK (Herrera-Pantoja et al., 2012). Indeed, in groundwater-fed wetlands across East Anglia, Herrera-Pantoja et al. (2012) found that a general projected decline in water levels could result in loss of species by the end of the 21st century. In the Elmley Marshes, southeast England, higher emissions scenarios were associated with lower groundwater and ditch water levels, and reductions in the magnitude and duration of surface inundation (Thompson et al., 2009). The range and complexity of impacts from climate change upon such wetlands leads to the need for assessment on an individual site basis in relation to a wetland's water supply mechanisms and position within the catchment (Acreman et al., 2007).

1.4.2 Assessing wetland ecological impacts of climate change

Hydrological changes due to climate change may be linked to water level requirements of different species and communities to infer ecological impacts (Acreman et al., 2009; Wheeler et al., 2004). As discussed in Section 1.3.2.1, water table level regime is a dominant control on wetland plant communities. Modifications to a wetland's hydrological regime may also be linked to changes in animal species distribution. As has also been discussed, focus has centred on indirect impacts to wading birds through the habitat requirements of their macroinvertebrate prey. Alterations to a wetland's water balance, and in turn its water level regime, due to climate change could lead to shifts in habitat availability (Johnson et al., 2005), and affect the capacity of a wetland to support populations of conservation importance (Herron et al., 2002; Sorenson et al., 1998; Thompson et al., 2009). In point, Thompson et al. (2009) showed that resultant ecological impacts from climate induced changes to the hydrological regime of the Elmley Marshes, southeast England, included the loss of grassland species and habitat for wading birds.

A framework to assess the ecohydrological impacts of climate change on wetlands was proposed by Acreman et al. (2009). This forms a tiered risk-uncertainty approach, starting with simple and generalised screening assessments, moving to complex models dependent on the detail and uncertainty involved. Tier 1 is termed 'risk screening', constituting a qualitative assessment of potential climate induced risks. Tier 2 comprises a generic quantitative assessment to ascertain possible consequences of climate change. Models used in this tier are not intended to represent specific sites, rather processes occurring over wider areas or regions. Tier 3 involves a detailed quantitative assessment to determine the magnitude of changes, the probability of any

consequences, and the level of risk for particular locations. Models for this tier are required that represent relevant processes at the site scale, such as coupled groundwater/surface-water modelling systems (see Section 1.3.5).

There are few hydrological modelling studies at a suitable resolution which link water table predictions directly to plant and animal requirements for individual wetlands (Carroll et al., 2015; Thompson et al., 2009). This was a key conclusion of the Whiteman et al. (2004) ecohydrological study into Great Cressingham Fen (see Section 1.3.5, Table 1-8). To the author's knowledge, none do so for individual wetlands with groundwater contributions. To assess the ecological impacts of climate change within wetlands, models are required that can accurately simulate groundwater levels at the fine-scale resolution associated with water level requirements of different species and communities (Thompson et al., 2009). Changes in water table level of less than 0.1 m may have profound effects on species composition, and provide conditions which favour distinct species or communities over those currently dominant at a given site (Wheeler et al., 2004). Whilst, as shown in Table 1-9, hydrological modelling has been used to assess some ecological effects of climate change, in many cases this has not been undertaken at a resolution sufficient to directly infer impacts for particular species and communities; instead surmising effects through changes in habitat availability (Barron et al., 2012; Candela et al., 2009; Johnson et al., 2005). Other studies have postulated impacts generalised over regional scales (Acreman et al., 2009; Herrera-Pantoja et al., 2012).

For surface water bodies, specifically watercourses, due to the direct relationship between habitat and flow (Section 1.3.2.1), hydraulic changes due to climate change may be linked to the depth and velocity requirements for different species and provide a measure of available physical habitat as a function of flow. The Physical Habitat Simulation (PHABSIM) system was the first modelling framework to quantify physical habitat for a particular discharge as a combined function of depth, velocity and substrate/cover (Bovee, 1978; Bovee, 1982; Bovee et al., 1998). The method is well suited to scenario analysis; the slope of the physical habitat-discharge relationship that forms a key output defines habitat sensitivity to change in particular flows (Figure 1-28). Becoming a legal requirement for many impact studies in the USA (Reiser et al., 1989), it is in standard use by the Environment Agency of England and Wales for determining sensitivity of rivers to abstraction, a requirement for Catchment Abstraction Management Strategies (Dunbar et al., 2002), and assessing ecological status for the European Water Framework Directive (Acreman et al., 2005). Despite criticisms of an insufficient link between habitat and biomass (Mathur et al., 1985; Orth and Maughan, 1986), models

Table 1-9. Summary of relevant studies using hydrological models to assess the impacts of climate change on wetland ecology

Source	Wetland type and location	Number / type of scenarios	Resolution / grid size	Ecological impacts
Johnson et al., 2005	Prairie potholes, central USA	3 / equilibrium scenario combinations of temperature and precipitation	Regional	Under a drier climate habitat for waterfowl would shift spatially
Acreman et al., 2009	Wet heaths / raised mires and riparian, various locations, UK	1 / UKCIP02 medium-high emissions scenario 2080s	Regional	Reduced summer rainfall and increased evaporation with put stress on plant communities in late summer and autumn
Barron et al., 2012	Coastal, Perth Basin, Western Australia	3 / outputs from 15 GCMs wet, medium and dry scenarios 2030s	500 m × 500 m	Impacts not uniform but could cause a threat to water-dependent ecosystems
Candela et al., 2009	Groundwater-fed, Majorca, Spain	2 / HadCM3 medium-high (A2) and medium-low (B2) emissions scenarios 2020s	250 m × 250 m	Aquifer recharge reduction could cause loss of wetland habitat
Herrera-Pantoja et al., 2012	Groundwater-fed fen, various locations, East Anglia, UK	1 / UKCIP02 high emissions scenario 2020s, 2050s and 2080s	250 × 250 m and 50 × 50 m	Decline in water levels could cause loss of species with small tolerance to dry conditions
Thompson et al., 2009	Lowland wet grassland, Elmley Marshes, southeast UK	4 / UKCIP02 low emissions, medium-low emissions, medium-high emissions and high emissions scenarios 2050s	30 × 30 m	Lower water levels result in loss of some grassland species and reduced suitability for wading birds such as lapwing and redshank
Carroll et al., 2015	Blanket bog, various locations, UK	1 / UKCP09 intermediate scenario 2011-2080	10 × 10 m	Falling water tables could cause 56-81 % declines in crane fly abundance, and 15-51 % declines in specialist predatory birds by 2051-2080

built on a similar concept have also been applied worldwide (Dunbar and Acreman, 2001), including RHYbasiM in New Zealand (Jowett, 1989), RSS in Norway (Killingtviert and Harby, 1994), EVHA in France (Ginot, 1995), HABIOSIM in Canada (Dunbar et al., 1997) and CASiMIR in Germany (Eisner et al., 2005; Jorde, 1996).

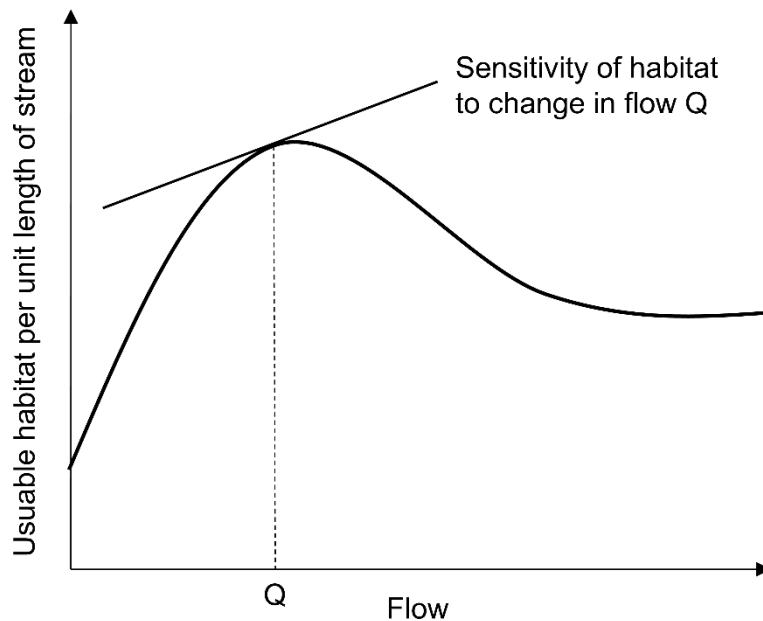


Figure 1-28. Habitat sensitivity to change in flow

In application, physical habitat modelling is site specific and resource intensive (Tharme, 2003), requiring extensive collection of field data at several different flows (Bovee, 1982) to obtain a physical habitat-discharge relationship. Approaches have been developed based on defining habitat-discharge relationships from fewer and/or simpler measurements of catchment, hydraulic or morphological characteristics. Lamouroux and Capra (2002) used non-linear mixed effect models on 58 French stream reaches to link within-reach changes to the Reynolds number, and between-reach changes to the Froude number at median daily discharge. A similar study showed the same result for over 100 New Zealand rivers (Lamouroux and Jowett, 2005). In the UK, strong relationships were shown between single measurements of channel form, river hydraulics and available habitat, based on 63 river reaches (Booker and Acreman, 2007; Klaar et al., 2014). However, these approaches are limited in success, restricted to region, and remain suited for screening exercises only. Any alterations to the river that affect the parameters of depth and velocity are not accounted for. These may include processes of instream macrophyte growth, groundwater exchange, or morphological adjustment that are key features of many dynamic systems.

The hydraulic component so often a feature of hydrological models merits application to physical habitat assessment through standard outputs of flow, depth and velocity. These enable calculation of velocity and depth profiles at each time step, and thus each distinct flow for a simulated period, rather than the few isolated measurements afforded by field surveys. A physical habitat-discharge relationship may be produced that is more representative of the range of flow and can incorporate dynamic processes modelled

within a river. Such a use of hydraulic models in assessing physical habitat availability demands fewer resources and may be readily applied to rivers through a range of scales and conditions. Yet, to the author's knowledge, there are no studies to this effect, despite worldwide import for evaluation of the impacts of environmental alterations, such as climate change, on fluvial ecosystems.

1.5 Summary

Wetlands provide numerous and vital environmental, cultural and economic functions and services. These services depend on the hydrological regime as it controls both biotic and abiotic processes. To maintain and in some cases maximise these services, wetland management is principally concerned with modifications to the hydrology, as well as vegetation and nutrients. In valley bottom wetlands, groundwater may exert a strong influence on the hydrological regime. In order to capture processes associated with groundwater/surface water interaction, techniques are required that can operate at a high spatial resolution. These may include temperature, which has not been comprehensively applied to wetlands before, and botanical indicators, as yet undefined in the UK.

Climate change will alter precipitation and evapotranspiration rates which, in turn, will result in changes to runoff and groundwater levels. The effects on regional aquifers and catchments may result in detrimental impacts to wetlands underlain by permeable geology, such as the chalk lowlands of southeast UK, where targeted assessments are needed. Water table level regime is a dominant control on wetland plant communities, whilst a direct relationship between physical habitat and flow means changes in river regime will affect aquatic biota. Alterations to a wetland's hydrological regime due to climate change could shift habitat availability and affect the capacity of a wetland to support populations of conservation importance.

An ability to predict the impacts of climate change is crucial for development of wetland management strategies. Models able to accurately represent wetland hydrology where processes are complicated by compound geology enable assessment of the effects of climate change. There are no hydrological modelling studies at a suitable resolution which link water table predictions directly to plant and animal requirements for individual wetlands with groundwater contributions. The hydraulic component of hydrological modelling systems enables assessment of physical habitat through standard outputs of flow, depth and velocity. Such a novel approach can incorporate dynamic processes such as macrophyte growth.

1.6 Aims and objectives

1.6.1 Aim

This thesis assesses the ecohydrological impacts of climate change on a riparian wetland in the chalk lowlands of the UK, with particular reference to the CEH River Lambourn Observatory.

1.6.2 Objectives

The objectives of the study are to:

1. Develop a conceptual model of the wetland through synthesis of existing research and further data acquisition obtained through monitoring and field surveys;
2. Develop a distributed hydrological/hydraulic model to simulate wetland hydrology under contemporary conditions and to produce a water balance of the site;
3. Project changes in hydrometeorological inputs to the hydrological/hydraulic model under scenarios of different climate sensitivities to incorporate the uncertainty associated with climate change;
4. Use the model to investigate how climate change scenarios affect wetland hydrology; and
5. Compare simulated hydrology under each climate change scenario with the requirements of species / communities for which the site is managed to assess ecohydrological impacts of climate change and resulting management implications.

1.7 Research design

Each of the objectives is associated with a number of sequential tasks necessary for their completion and, ultimately, to meet the overall aim of the study (Figure 1-29). The approach corresponds to a third tier investigation in the Acreman et al. (2009) framework to assess climate change impacts on wetlands (see Section 1.4.2). The consequences of climate change are examined in detail to quantitatively evaluate the ecohydrological repercussions to a chalk valley bottom, riparian wetland. This follows a programme of work which defines the structure of the thesis.

A desk study collates and reviews information already available into a site description of the River Lambourn Observatory and forms Chapter 2. Details of catchment characteristics are included, along with spatial examination of historical maps and

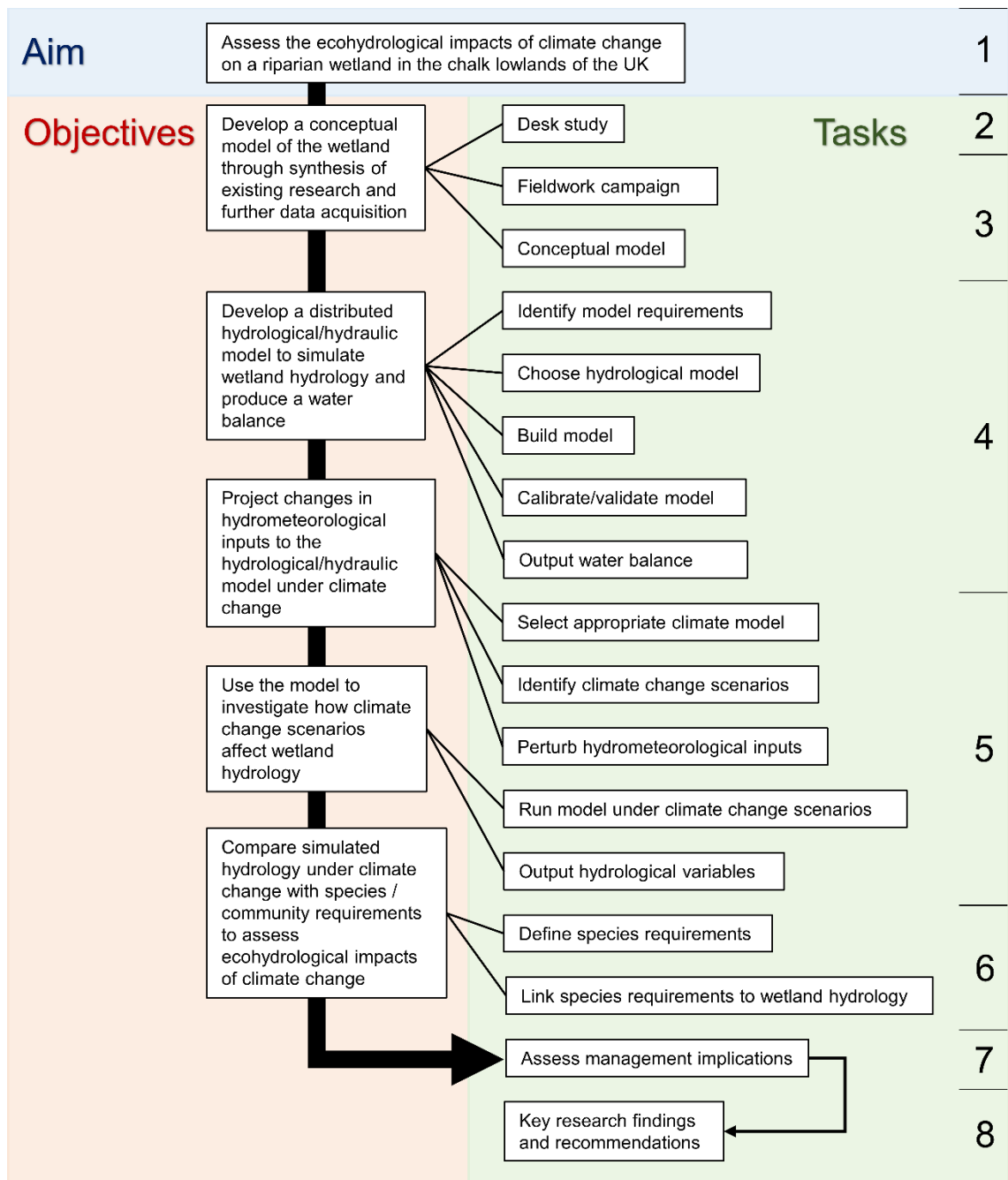


Figure 1-29. Research design showing aim, objectives, tasks and chapter numbers

photographs, and outputs of previous studies on the site, its catchment and the immediate surrounding location. Existing knowledge of topography, geology, hydrology, ecology and site management are brought together to form an initial conceptual model of the sites hydrological functioning. This aids the identification of areas that require further attention to improve understanding. A subsequent programme of field measurement and monitoring is presented in Chapter 3, culminating in a revised conceptual model. Site meteorology, channel morphology and groundwater levels are assessed, while the topographical data is improved. The issue of delineating areas of groundwater/surface water interaction is addressed.

The choice of a hydrological model based on requirements from the conceptual model, along with its ensuing development, calibration and validation are described in Chapter 4. There is a particular focus on groundwater/surface interaction and its effect on the water balance. The use of this model to investigate the hydrological impacts of climate change are investigated in Chapter 5. Climate change scenarios are derived from the UK Climate Projections 2009 (Murphy et al., 2009) ensemble of models for the 2080s. Projected changes to simulated wetland hydrology are then linked in Chapter 6 to the ecological requirements for plants and animals that are managed at the site. The ecological impacts of climate are assessed while management implications are discussed in Chapter 7. The concluding Chapter 8 highlights the main findings and implications of the research, along with its limitations. Recommendations for further research at the Observatory and in similar environments are provided.

1.8 Outputs

In addition to this thesis a number of outputs, in the form of journal publications, have already been derived from this research. At the time of thesis submission (April 2016), three have been published whilst one further paper is under review:

- House, A.R., Thompson, J.R., Roberts, C., de Smeth, K., Old, G. & Acreman, M.C. (in review). Projecting impacts of climate change on habitat availability in a macrophyte dominated chalk river. *Ecohydrology*.
- House, A.R., Thompson, J.R. & Acreman, M.C. (2016). Projecting impacts of climate change on hydrological conditions and biotic responses in a chalk valley riparian wetland. *Journal of Hydrology*, 534: 178-192. DOI: 10.1016/j.jhydrol.2016.01.004.
- House, A.R., Thompson, J.R., Sorensen, J.P., Roberts, C. & Acreman, M.C. (2015). Modelling groundwater/surface-water interaction in a managed riparian chalk valley wetland. *Hydrological Processes*. DOI: 10.1002/hyp.10625.
- House, A.R., Sorensen, J.P., Goody, D.C., Newell, A.J., Marchant, B., Mountford, J.O., Scarlett, P., Williams, P.J. & Old, G.H. (2015). Discrete wetland groundwater discharges revealed with a three-dimensional temperature model and botanical indicators (Boxford, UK). *Hydrogeology Journal*, 23(4): 775-787. DOI: 10.1007/s10040-015-1242-5.

Chapter 2

The CEH River Lambourn Observatory

2.1 Introduction

The Centre for Ecology and Hydrology (CEH) River Lambourn Observatory, a Site of Special Scientific Interest (SSSI) and Special Area of Conservation (SAC), is the research site for this study. In this chapter existing information available about the site and its catchment are collated into a desk study. Details of site history and management, ecology, topography, geology and hydrology are reviewed to provide an account of the existing state of understanding. This allows identification of areas where additional knowledge is required and further investigation warranted.

The CEH River Lambourn Observatory (51.445° N 1.384° W) encompasses c.10 ha of riparian wetland bordering a 600 m reach of the River Lambourn (Figure 2-1). The Observatory is located in the village of Boxford, West Berkshire, UK, at a conspicuous bend in the River Lambourn, which drains the Chalk of the Berkshire Downs (Figure 2-2).

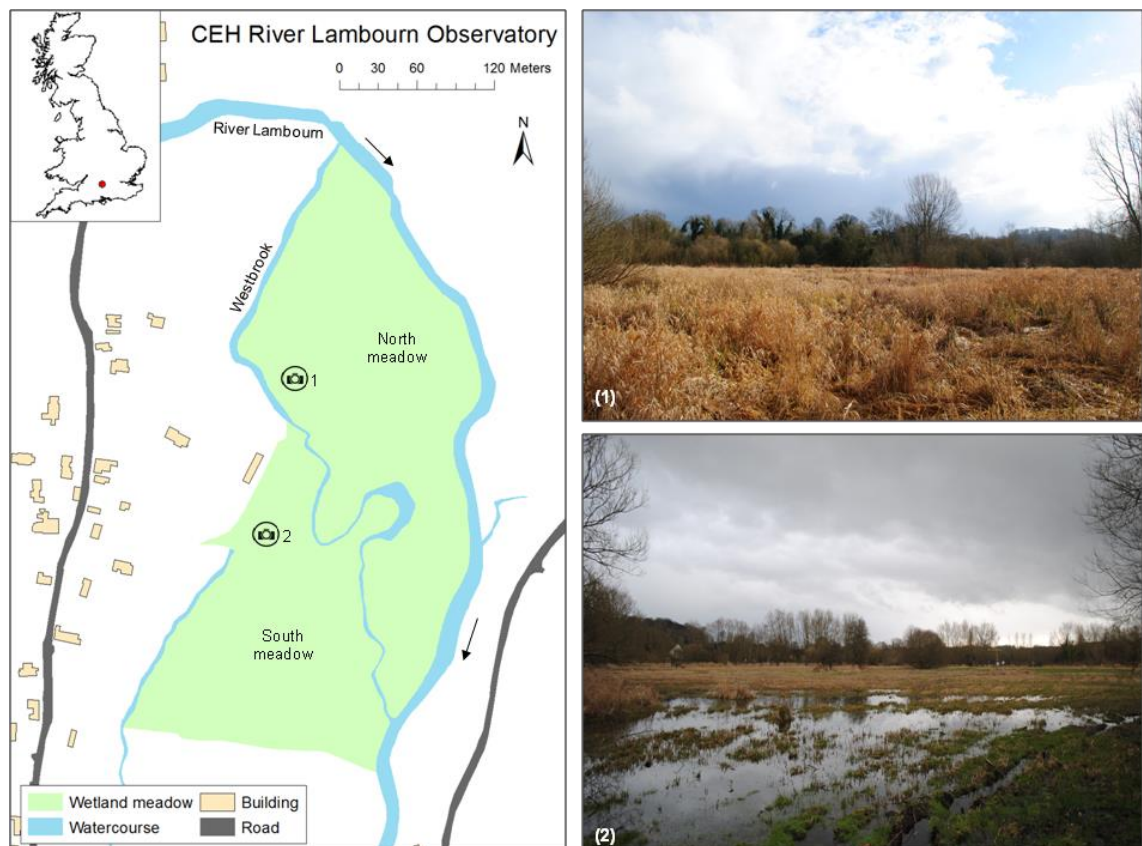


Figure 2-1. The CEH River Lambourn Observatory, with southeast facing photos of the north (1) and south (2) meadows.

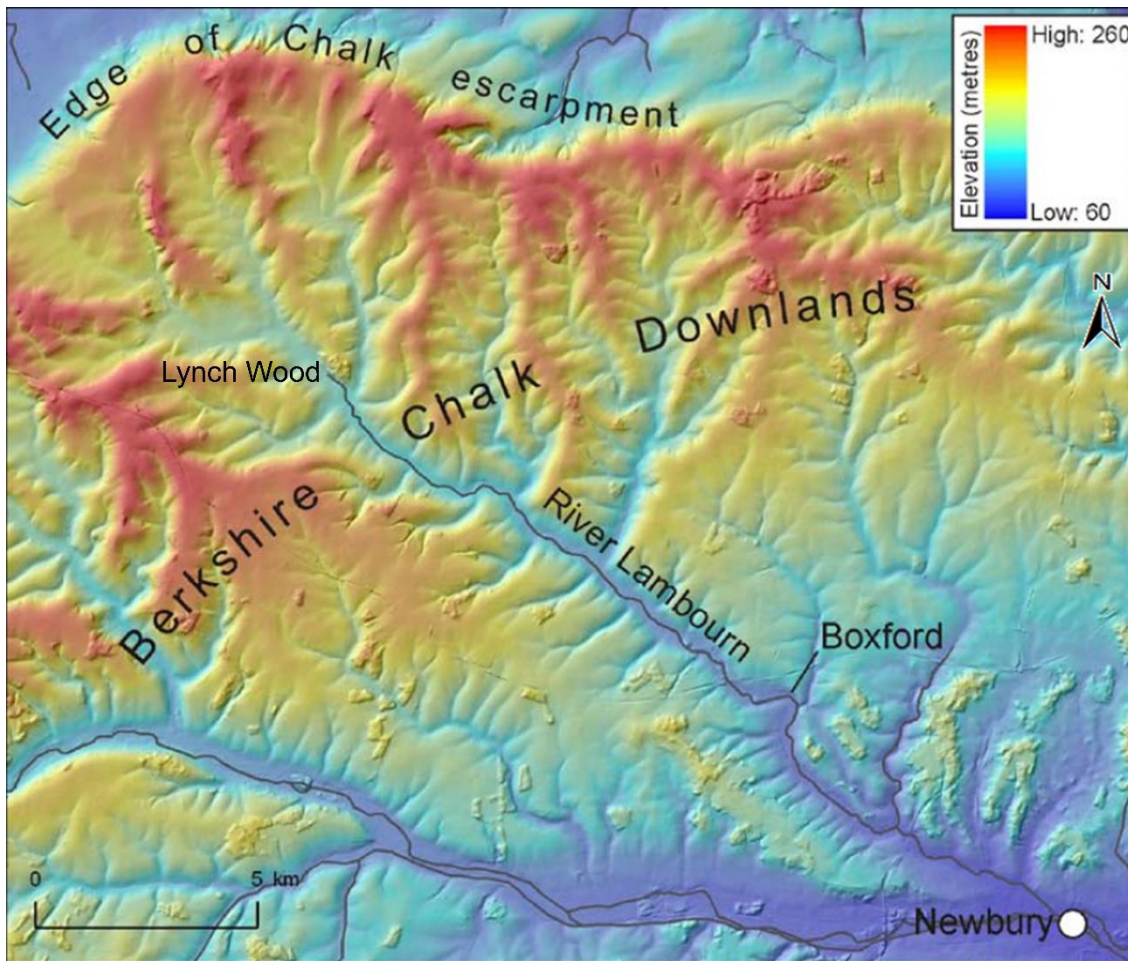


Figure 2-2. Location of the Boxford area and River Lambourn within the Berkshire Downs (after Allen et al., 2010). NEXTMap Britain elevation data from Intermap Technologies. OS data ©Crown Copyright. All rights reserved. BGS 100017897/2009

The ephemeral source of the River Lambourn is 13 km upstream at Lynch Wood, Lambourn (51.512° N, 1.529° W), the perennial head of which is situated 6-7 km downstream of source at Maidencourt Farm (51.481° N, 1.464° W) (Figure 2-3). The catchment area at the Observatory is 162 km². To the west of the river, the wetlands are divided by the Westbrook Channel into a northern and southern meadow. A network of shallow channels, remnants of a former system, cross the wetlands (Figure 2-4).

The wetland and River Lambourn are designated as a SSSI and SAC due to their importance for *Vertigo moulinsiana* (Desmoulin's whorl snail), *Lampetra planeri* (brook lamprey) and *Cottus gobio* (bullhead), plus certain habitats (Annex 1 habitat from EU Habitat Directive: Water courses of plain to montane levels with *Ranunculum fluitantis* and *Callitriche-Batrachion* vegetation) and terrestrial plant communities (MG8 vegetation community of the UK National Vegetation Classification; Rodwell, 1991).

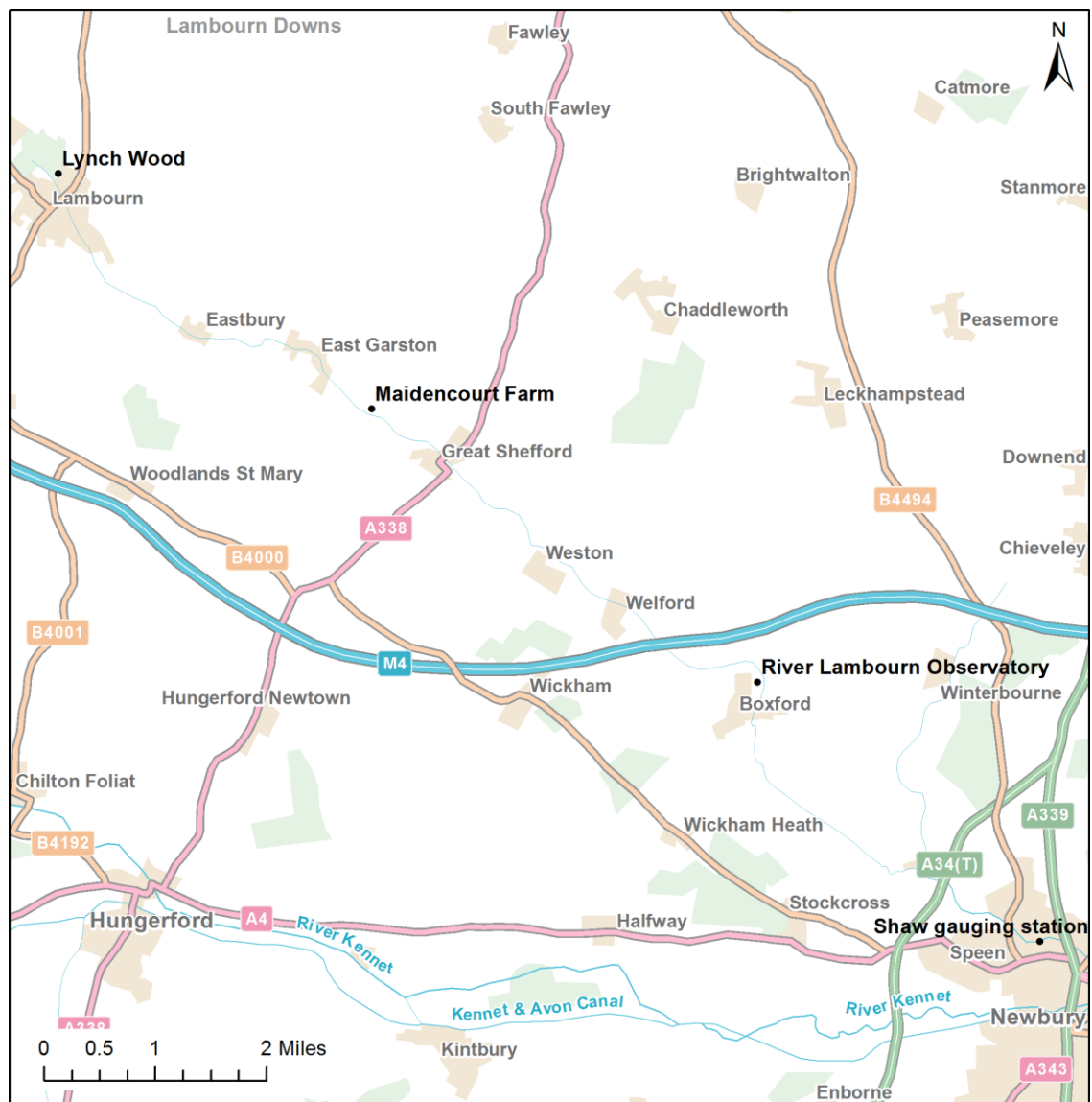


Figure 2-3. Locations of the River Lambourn Observatory, ephemeral source of the River Lambourn at Lynch Wood, perennial head at Maidencourt, Shaw gauging station, and surrounding area. OS data ©Crown Copyright

A certain amount of research has been conducted at the Lambourn Observatory and surrounding area. An adjacent site, situated about 100 m upstream of the Observatory's northern boundary, has been the focus of substantial study (see Figure 2-8). Initially set up under the NERC Lowland Catchment Research (LOCAR) thematic programme (Wheater and Peach, 2004), it formed one of several sites selected to improve the science required to support management needs for permeable lowland catchments. Due to its proximity, the findings provide a useful parallel and starting point for conceptualisation of the wetland. In the Observatory itself, Atkins (2005) conducted a hydrological study as part of an impact assessment into the Lambourn floodplain SAC, and Musgrave (2006) investigated water sources to help improve understanding of

floodplain wetlands. A separate study by Musgrave and Binley (2011) in a small wetland area to the east of the Observatory used ground penetrating radar and electrical resistivity tomography to investigate groundwater-surface water interaction. More recently, studies into the riparian and instream effects of weed cutting in the River Lambourn (Old et al., 2014), and a geophysical investigation of the wetland (Chambers et al., 2014) have resulted in key conceptual findings, discussed in this chapter.

2.2 History and management

The Observatory, as with much of the floodplains of the River Lambourn and nearby River Kennet (Figure 2-3), were managed as flood pastures and water meadows until the middle to late 20th century (Everard, 2005). Historic maps dating back to the 1880s show a network of predominantly linear conduits, sluices and aqueducts, the relics of which are still apparent (Figure 2-4). These would have been used for controlled winter flooding, known as 'floating', to irrigate the meadows, protecting them from frost and encouraging early hay crop growth in the spring (Everard, 2005).

Water meadows arose out of 16th and 17th century agricultural innovations, yet before modern agricultural intensification (Everard, 2005). They should not be confused with grazing pastures, where a network of drainage ditches partially controls flooding, or flood meadows, which are simply inundated with flood water. Prior to this it is difficult to determine how the Observatory would have appeared, although it is likely it would have been managed as far back as the Iron or Bronze Age. Prehistorically, much of the area would have been covered by fen overgrown with trees, known as 'carr'.

Grazing gradually came to an end on the water meadows from the mid-1960s to the 1980s. This is reflected in the vegetation succession from swamp and fen in the north to remnants of the MG8 community in the south. Most of the old channels are naturally infilled and absent from current maps. Recent (1990s) modifications include a flight pond for shooting game, and a dug out recreational pond with sluice, both on the Westbrook (Figure 2-5).

In autumn 2007, CEH purchased the site to allow for long term research on rivers and their floodplains in southern England. A monitoring programme at the Observatory, set up in collaboration with the British Geological Survey (BGS), was intended to bring together and consolidate cross-disciplinary expertise within the fields of hydrogeology, hydrology, hydrochemistry, sedimentology and ecology. Research at the site enables integrated study of the relationship between geology, groundwater, surface water and river ecology. Current management is, so far, confined to the river, where under the

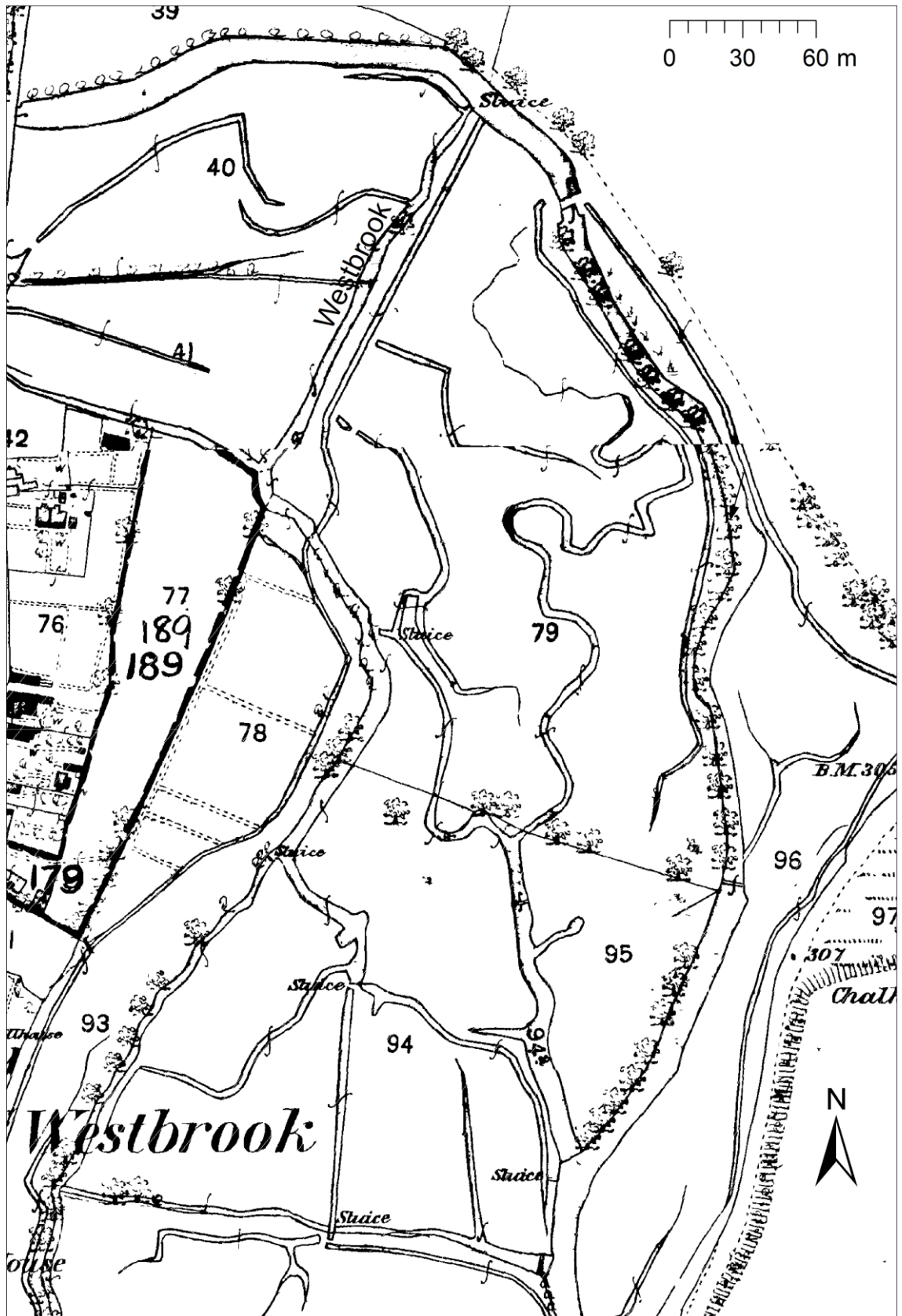


Figure 2-4. 1st edition 1880s historic map of the River Lambourn Observatory and surrounding area © Crown Copyright and Landmark Information Group Limited 2015. All rights reserved.

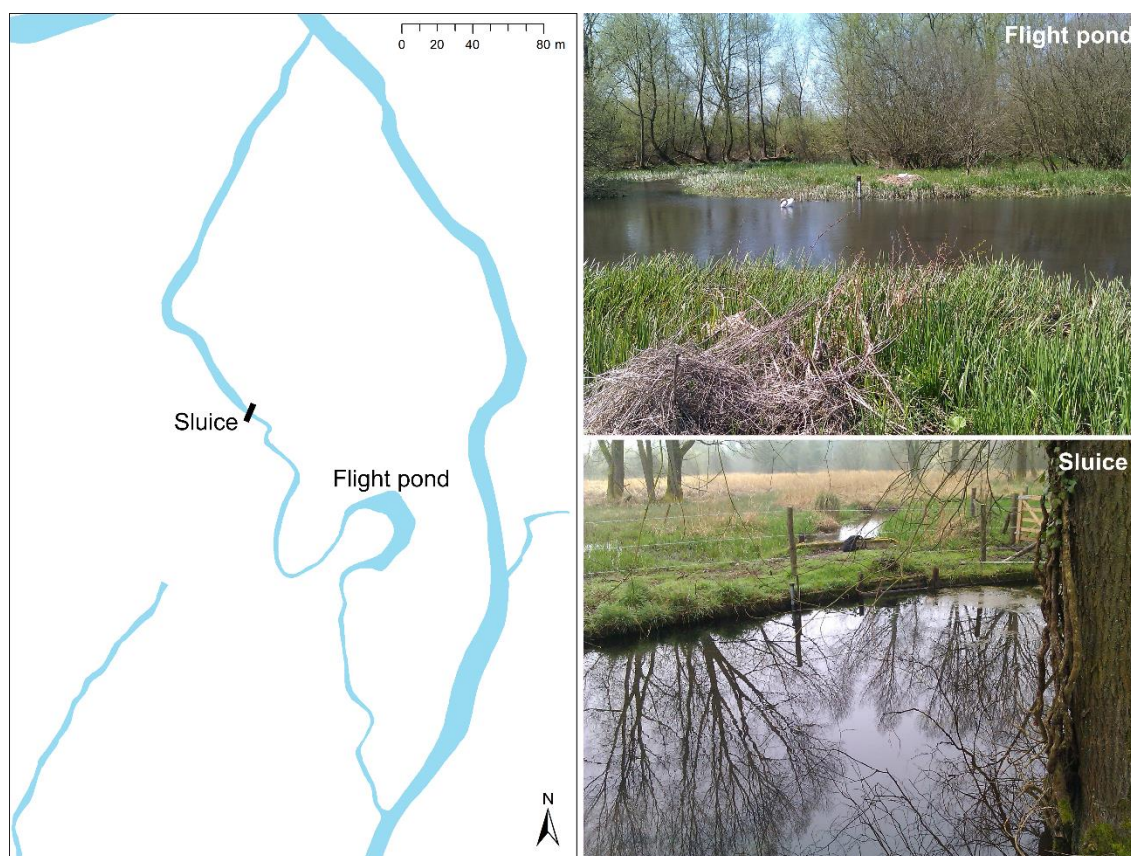


Figure 2-5. Locations and photos of the Westbrook flight pond and sluice

Flood and Water Management Act 2010, CEH must fulfil legal obligations as the riparian landowner to maintain the conveyance of river water through the site to manage flood risk. Hence, instream macrophyte growth is cut back periodically to maintain flood conveyance and lower water levels, yet also to maintain fisheries for brown trout (Old et al., 2014).

2.3 Ecology

A total of 17 species of grass, seven species of sedge and 76 species of grassland herb have been recorded at the Observatory, whilst the insect fauna is also diverse, with species rare in Berkshire including the beetles *Cantharis pallida* and *Subcoccinella 24-punctata* and the bug *Neophilaenus campestris* (Natural England, 1986). The wetland vegetation reflects a succession, with plant communities classed under the NVC system (Rodwell, 1991) grading from S7 *Carex acutiformis* swamp on the northernmost meadows towards remnants of a more typical MG8 *Cynosurus cristatus*-*Caltha palustris* flood pasture southwards towards the drier areas of the site (Natural England, 2012). Patches of *Alnus glutinosa* (alder) and *Salix caprea* (sallow) scrub also occur sporadically across the SSSI. The north meadow contains species characteristic of the S7 community, including *Carex acutiformis* (lesser pond-sedge), *Galium palustre*

(common marsh-bedstraw), *Mentha aquatica* (water mint), *Valeriana dioica* (marsh valerian), *Angelica sylvestris* (wild angelica), *Filipendula ulmaria* (meadowsweet), *Equisetum palustre* (marsh horsetail) and *Arrhenatherum elatius* (false oat-grass). Lesser pond-sedge and *Glyceria maxima* (reed sweetgrass) dominate, while *Carex paniculata* (greater tussock sedge) and *Sparganium erectum* (branched bur-reed) are prominent. In the south meadow remnants of the MG8 community are present in species such as *Caltha palustris* (marsh marigold), meadowsweet, *Ranunculus repens* (creeping buttercup), *Ranunculus acris* (meadow buttercup), *Eleocharis palustris* (common spike-rush), and *Poa trivialis* (rough meadow-grass).

Aside from *Lampetra planeri* (brook lamprey) and *Cottus gobio* (bullhead) for which the site is designated, there are four species of pelagic fish present at the site: brown trout (*Salmo trutta*), grayling (*Thymallus thymallus*), 10-spined stickleback (*Pungitius pungitius*) and 3-spined stickleback (*Gasterosteus aculeatus*). The macrophyte community is dominated by water crowfoot (*Ranunculus* spp. *pseudofluitans* mixed with smaller quantities of *R. penicillatus* spp. *pseudofluitans* × *Ranunculus peltatus* hybrid). Frequent patches of water starwort (*Callitriche* spp.), water parsnip (*Berula erecta*) and watercress (*Rorippa nasturtium-aquaticum*) also occur.

The meadows once supported breeding *Gallinago gallinago* (snipe) (Everard, 2005), although these are no longer present and a general decline in the UK and Europe has been linked to losses in lowland wet grassland (Ausden et al., 2001). Formerly (in the 1980s) the site was also a refuge of *Austropotamobius pallipes* (freshwater crayfish), although these became extinct with the invasion of *Pacifastacus leniusculus* (North American signal crayfish). A survey undertaken in 2012 has also shown a reduced presence of Desmoulin's whorl snail (Figure 2-6), the species which contributes to the site's scientific and nature conservation status, suggesting a gradual decline since the 1990s (Natural England, 2012).

2.4 Topography

A topographic survey of the Observatory, undertaken at the end of May 2012 indicates a rather flat landscape, to be expected given the nature of the site (Figure 2-7) (Chambers et al., 2014). Although not spatially comprehensive, the survey used differential GPS (dGPS) to georeference 2815 locations with an approximate grid resolution of 5 x 5 m. Points were taken at noticeable changes in slope or, otherwise, 1 m apart. Dense scrub and watercourse boundaries confined the survey extent.



Figure 2-6. Desmoulin's whorl snail (*Vertigo moulinsiana*) © Roger Key

Elevations were interpolated using a two-dimensional minimum curvature spline technique to maintain input values.

Height variation over the site was found to be little more than 1 m. Elevations ranged from 89.74 to 91.33 mAOD, with mean elevation at 90.49 mAOD. A gentle north-south slope of 1.6% follows the valley bottom. The relic channel network which crosses the wetland, following the gradient, is noticeable in the form of linear, branching features.

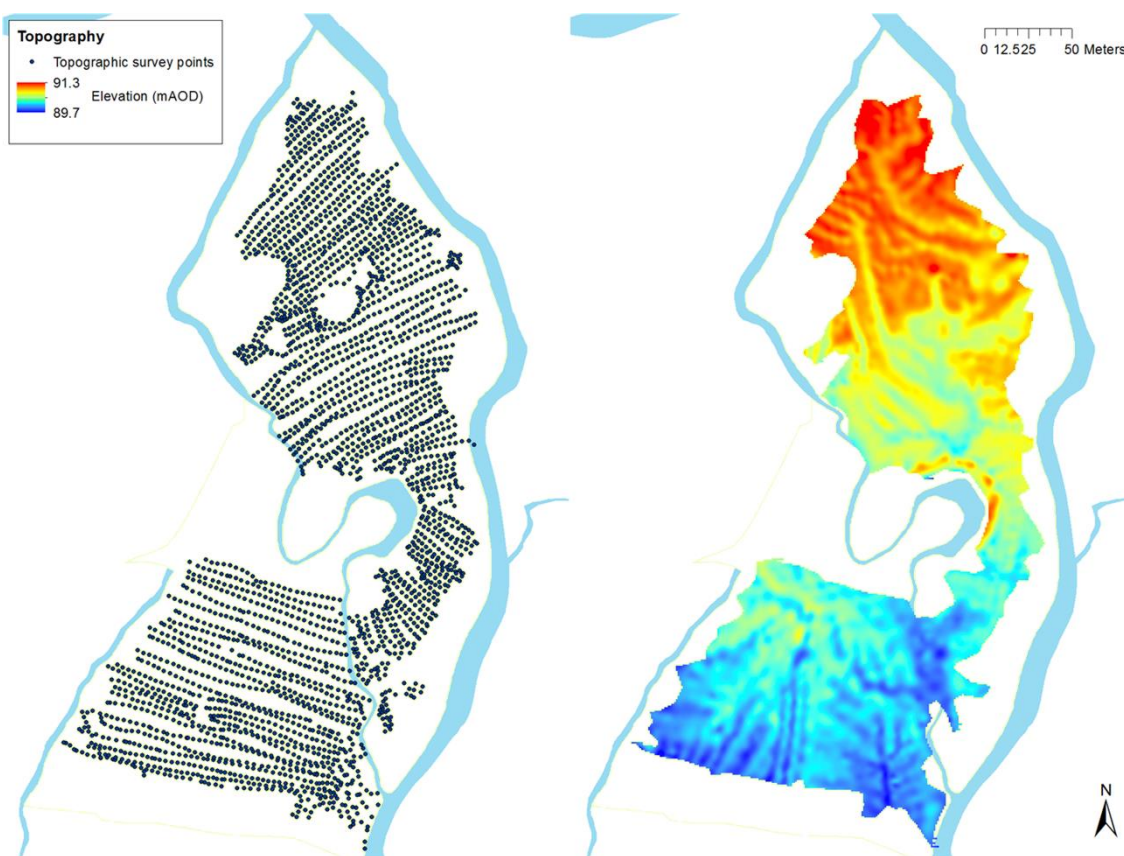


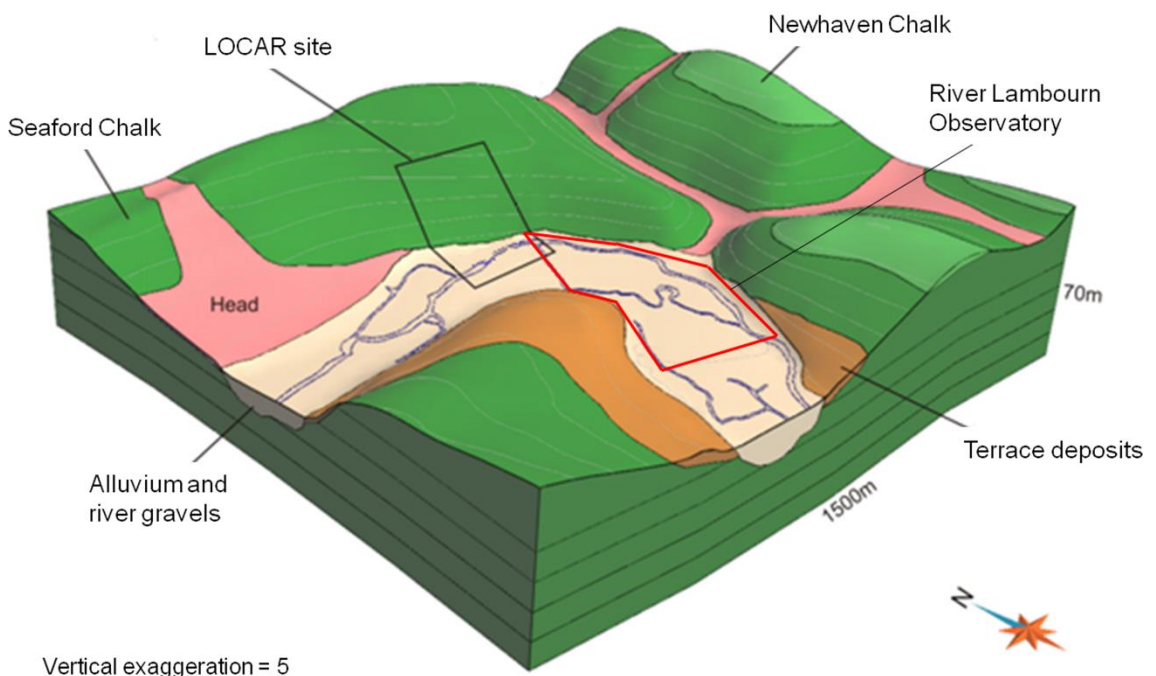
Figure 2-7. Topographic survey points and resulting elevations for the Observatory.

2.5 Geology

2.5.1 The Berkshire Downs and Lambourn Valley

The Chalk Group of the Berkshire Downs can be up to 252 m thick and dips south easterly at 1-2° away from its northern escarpment boundary (Aldiss et al., 2002). The River Lambourn generally follows this gradient, possibly controlled by joints in the Chalk. Palaeogene deposits and superficial drift from the Quaternary period overlay the Chalk. These comprise Clay-with-flints on interfluvial and river terrace deposits, head and alluvium on valley floors (Aldiss et al., 2010). A thin discontinuous layer of highly weathered and low permeability 'putty chalk' occurs in interfluvial on the upper surface of the Chalk (Younger, 1989). Formed from periglacial freeze-thaw action, the putty chalk layer may be found patchily throughout the Thames and tributary valleys.

At Boxford, the River Lambourn cuts into the Seaford Chalk Formation (Figure 2-8), a uniform soft to medium-hard chalk with frequent flint nodules, which dips at 1-2° to the southeast across the valley (Woods and Aldiss, 2004). Boreholes drilled at the LOCAR site showed up to 7.5 m of river terrace deposits and alluvium overlying the Chalk (Allen et al., 2010). These are mostly coarse-grained gravels with typically 50% of clasts in the 25–100 mm size range. Sand, silt and clay do not normally make up more than 5% of the deposit, although thin beds of sandy gravel occur locally. The gravels are generally



Vertical exaggeration = 5

Figure 2-8. Block geology of the Boxford research sites (after Allen et al., 2010). NEXTMap Britain elevation data from Intermap Technologies. OS data © Crown Copyright. All rights reserved. BGS 100017897/2009.

3-4 m thick, with local thickening and thinning suggesting an uneven erosion surface at the top of the Chalk. Composition is primarily of rounded flint clasts, although the basal 1-2 m can include a high proportion of reworked chalk material which may have resulted from downcutting into the river terrace deposits. An additional approximately 5 m thick layer of structureless putty chalk exists between the gravel-chalk interface and coherent chalk bedrock.

The term 'gravels' is used hereafter to describe the mainly coarse-grained lithologies which form the river terrace deposits. The gravels underlie a heterogeneous layer of peat, sands, silts, clays and tufa 1-2 m thick (Allen et al., 2010), the surface of which forms the present floodplain. These peaty alluvial deposits, subsequently termed 'peat', meet the undulating gravel surface abruptly. Localised thickening occurs in narrow and linear depressions that may have resulted from infilling of relic channels in the gravels. At the valley sides these deposits merge with clay-rich head and slope-wash deposits.

Downstream fluvial stratigraphy of the River Kennet (Collins et al., 2006) provide clues as to the development of the River Lambourn and its deposits. The chalk and flint gravels were most likely deposited during the periglacial Late Pleistocene (Devensian) in high-energy bed load channels (Murton and Belshaw, 2011). The flow regime was dominated by snowmelt with high spring and summer runoff (Collins et al., 2006). Permafrost would have limited infiltration to the permeable chalk bedrock. An extensive network of dry valleys through the Berkshire Downs indicate the once extensive high-energy river network. The peat and alluvial cover accumulated during the Holocene, when the temperate climate produced low-energy rivers and wetlands. These were probably mostly groundwater fed from the Chalk aquifer.

2.5.2 Intrusive exploration at the River Lambourn Observatory

In the Observatory wetland a peat depth survey was carried out in conjunction with the 2012 topographic survey (Figure 2-9), georeferenced at the same 2815 locations (Chambers et al., 2014). Depth was determined by inserting a 6 mm diameter steel rod to contact between the penetrable peat and impenetrable gravels. Drilling was also undertaken in November 2012 using a Dando Terrier™ percussion drilling rig at three locations across the site. Locations were influenced by accessibility and site conditions. Cores were recovered with a hollow stem auger in U100 tubes.

Peat depths ranged from 0.39 to 1.86 m below ground level (bgl), with a mean of 0.89 m (Figure 2-9i). The peat depth was in excess of that measurable at six locations where it was assumed to be 1.86 m deep. The peat thins to the southeast and east of the survey

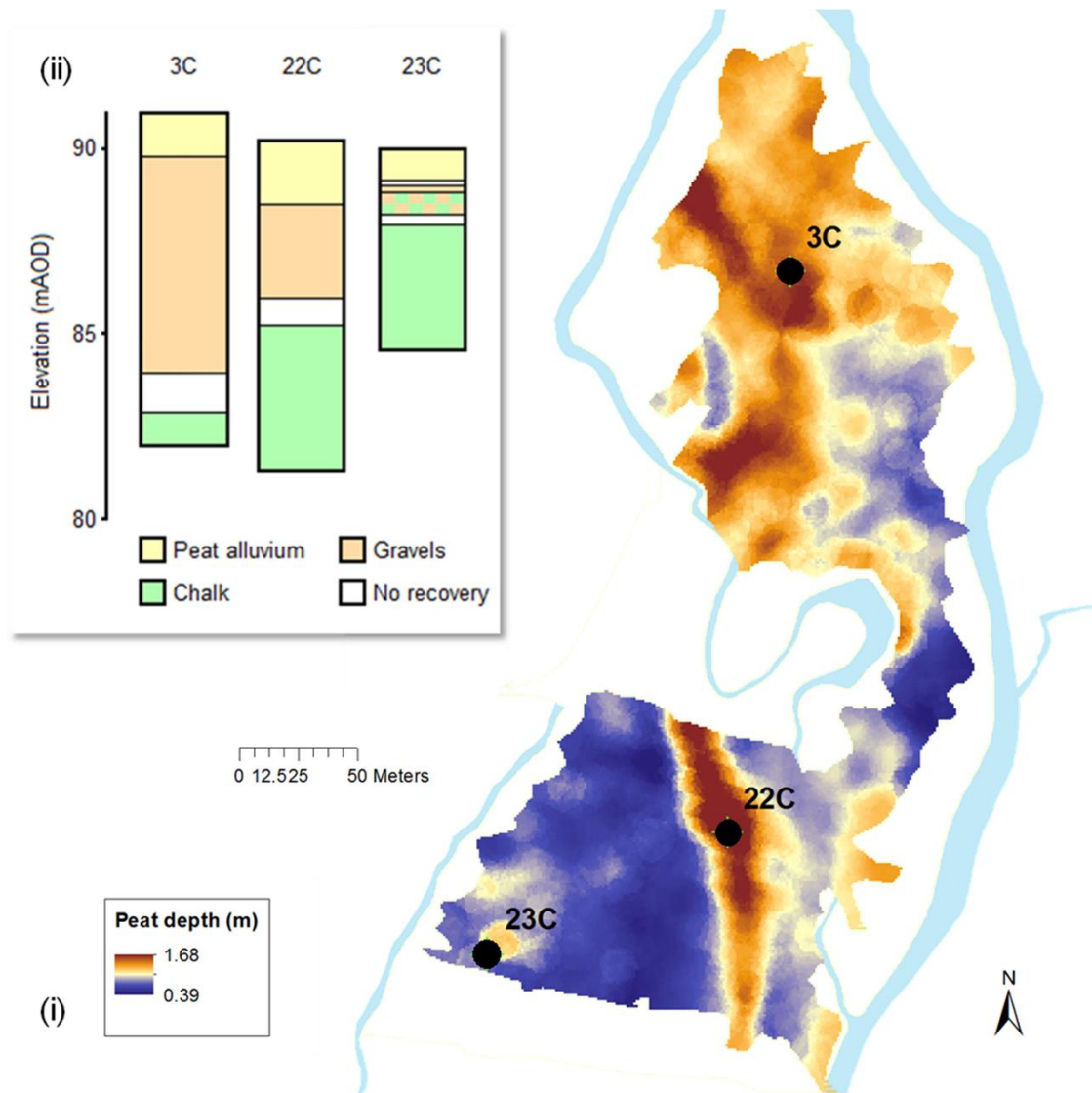


Figure 2-9. Intrusive geological exploration: (i) Observatory peat depth below ground level and borehole positions, (ii) borehole logs

extent. A channel of thicker peat runs north to south across the site, although it is thicker and broader in the north. This covers a topographic low in the surface of the gravels.

Boreholes 3C and 22C were sited to target the peat channel in the north and south meadows, respectively (Figure 2-9ii). A thick layer of peat is shown overlying several metres of gravel and then chalk bedrock. Coarse material at the gravel-chalk interface caused core losses and impeded drilling. Towards the southeast tip of the site, borehole 23C revealed an approximately 1 m layer of peat overlying a thin band of sand and gravel (<0.2 m), a mixture of chalk and gravel, and chalk bedrock. In all boreholes the peat alluvium contained thin layers of clay, observed between 0.1 and 0.3 m depth. The chalk became firmer with depth, being very soft and highly weathered at the top, consistent with the putty chalk described by Younger (1989).

2.5.3 Geophysical investigation at the River Lambourn Observatory

A non-invasive geophysical technique was employed in 2012 to help further reveal subsurface architecture (Chambers et al., 2014). This combined three-dimensional electrical resistivity tomography (ERT) and the use of isosurface based edge detectors. ERT uses arrays of electrodes at ground surface or within boreholes to measure potential differences resulting from applied currents. An inverse problem is solved to provide a model of the subsurface resistivity distribution (Loke et al., 2013). Interfaces between geological layers are represented in the inverted ERT image by a resistivity isosurface. Independent resistivity measurements from boreholes are used to ascertain the isosurface value within the resistivity model. The surface may then be interpolated through the model. As a single resistivity value determines the surface, the method is valid only where a clearly layered structure exists.

The 3D ERT surveys were executed over approximately 1.5 ha for the south meadow in April 2012 and 1.6 ha for the north meadow in December 2012. Electrode placements for the survey are presented in Figure 2-10, along with lines of vertical cross-sections through the resultant model.

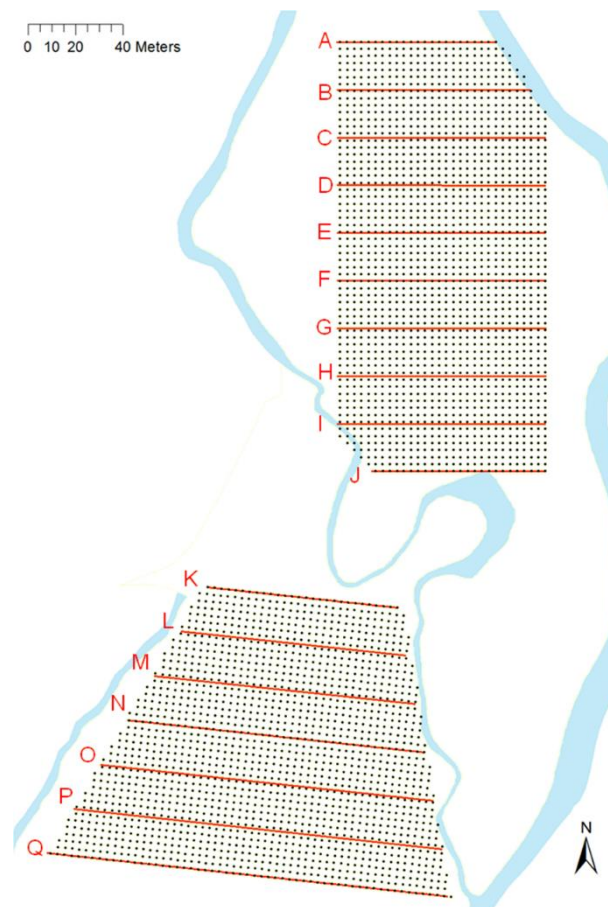


Figure 2-10. ERT electrode positions and lines of vertical cross-sections.

The 3D model of both north and south ERT survey results reflects the lithostratigraphy found in boreholes 3C, 22C and 23C. Vertical cross-sections through the model are displayed in Figures 2-11 and 2-12. Low resistivity peats are shown to span both meadows in the form of a channel, coincident with intrusive investigations. Thinner peat (<0.9 m) in other areas is not well resolved in the model. High resistivity gravels are thicker and more continuous in the north meadow, nearer to the course of the river. In the south, the gravels are more variable and thin to the west, almost disappearing towards the southwest boundary, which is in agreement with the log from borehole 23C. Variability in the gravel resistivities will be due to variations in porosity and fines content (silt and clay particles). Chalk resistivities, between those of the peat and gravels, vary substantially. Resistivities generally increase with depth, which is accepted to relate to

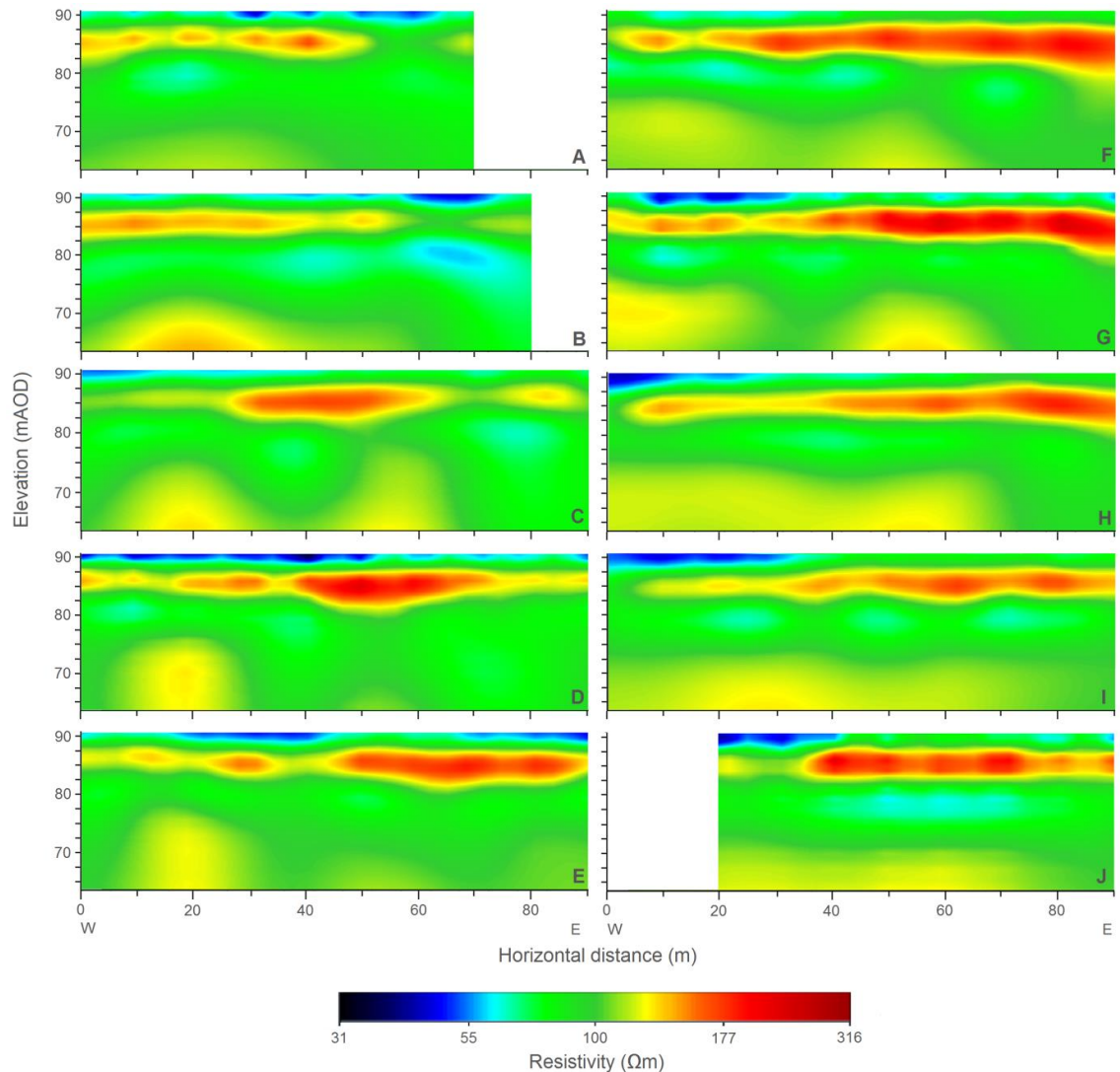


Figure 2-11. Vertical cross-sections of resistivity in the north meadow.

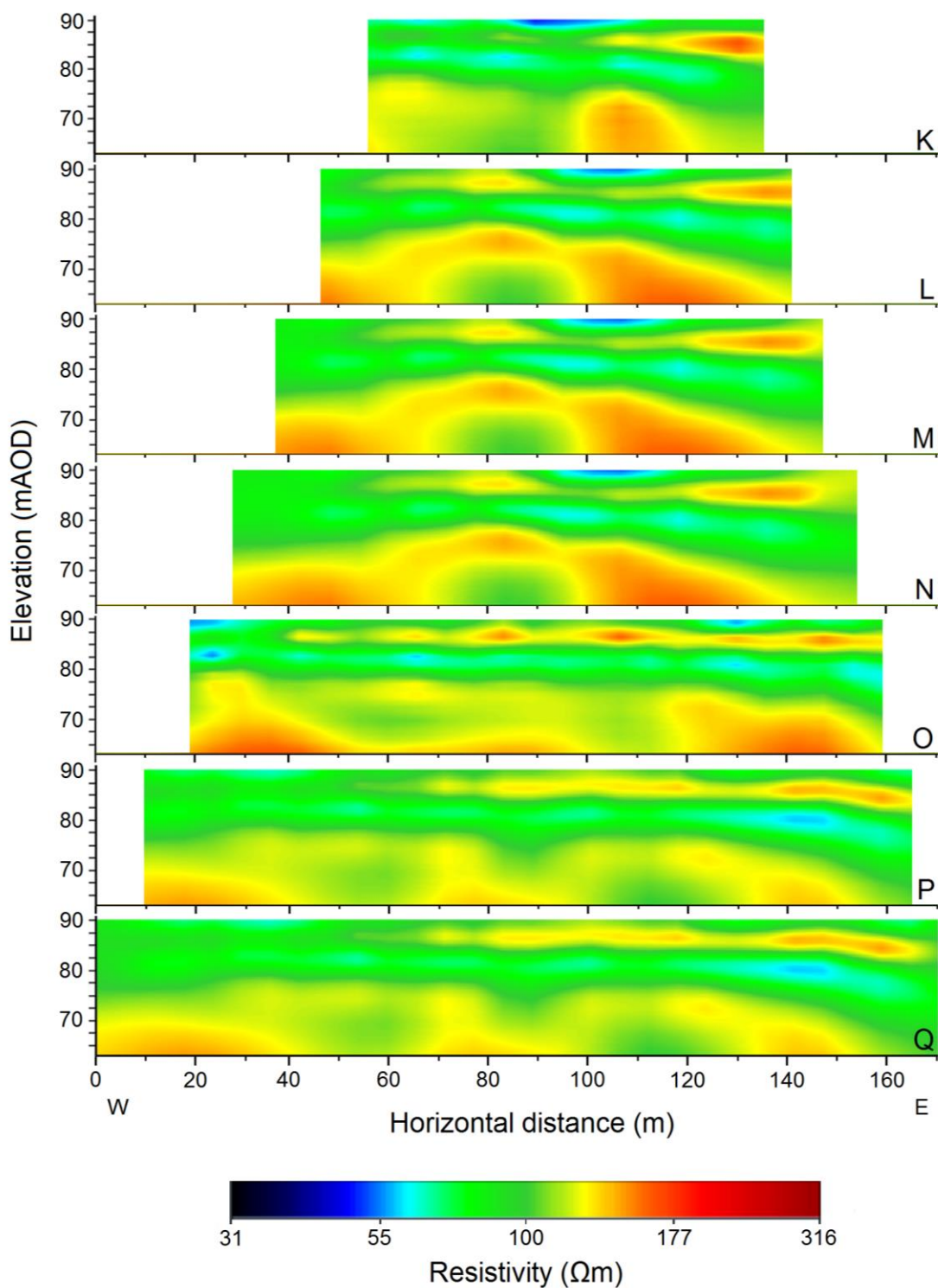


Figure 2-12. Vertical cross-sections of resistivity in the south meadow.

weathering. A weathered zone may be seen in the model as a layer of highly variable thickness (1-10 m) at the top of the Chalk, with resistivities <75 Ωm.

The weathered zone is consistent with the formation of clay-rich, and hence electrically conductive, putty chalk (Younger, 1989), as found when drilling. Below the weathered

zone, changes in resistivity are likely a product of fractures and the occurrence of flints. Younger (1989) proposed a model for the formation of putty chalk beneath river valleys for the nearby Reading area of the Thames Valley. Annual freeze-thaw from periglacial conditions causes pulverisation of Chalk beneath minor channels. Formations of putty chalk are thus produced at the interface between the gravel and the Chalk. This supports the highly variable distribution of the putty chalk through the survey area.

Interfaces between subsurface layers are displayed in Figure 2-13. The isosurface for the peat-gravel interface was limited to peat thicknesses >0.9 m due to model resolution. However, the peat channel can be clearly distinguished running north to south, with a broader distribution in the north (Figure 2-13i). The interface also reveals that the peat sits within topographic depressions in the surface of the gravels. The gravel-Chalk

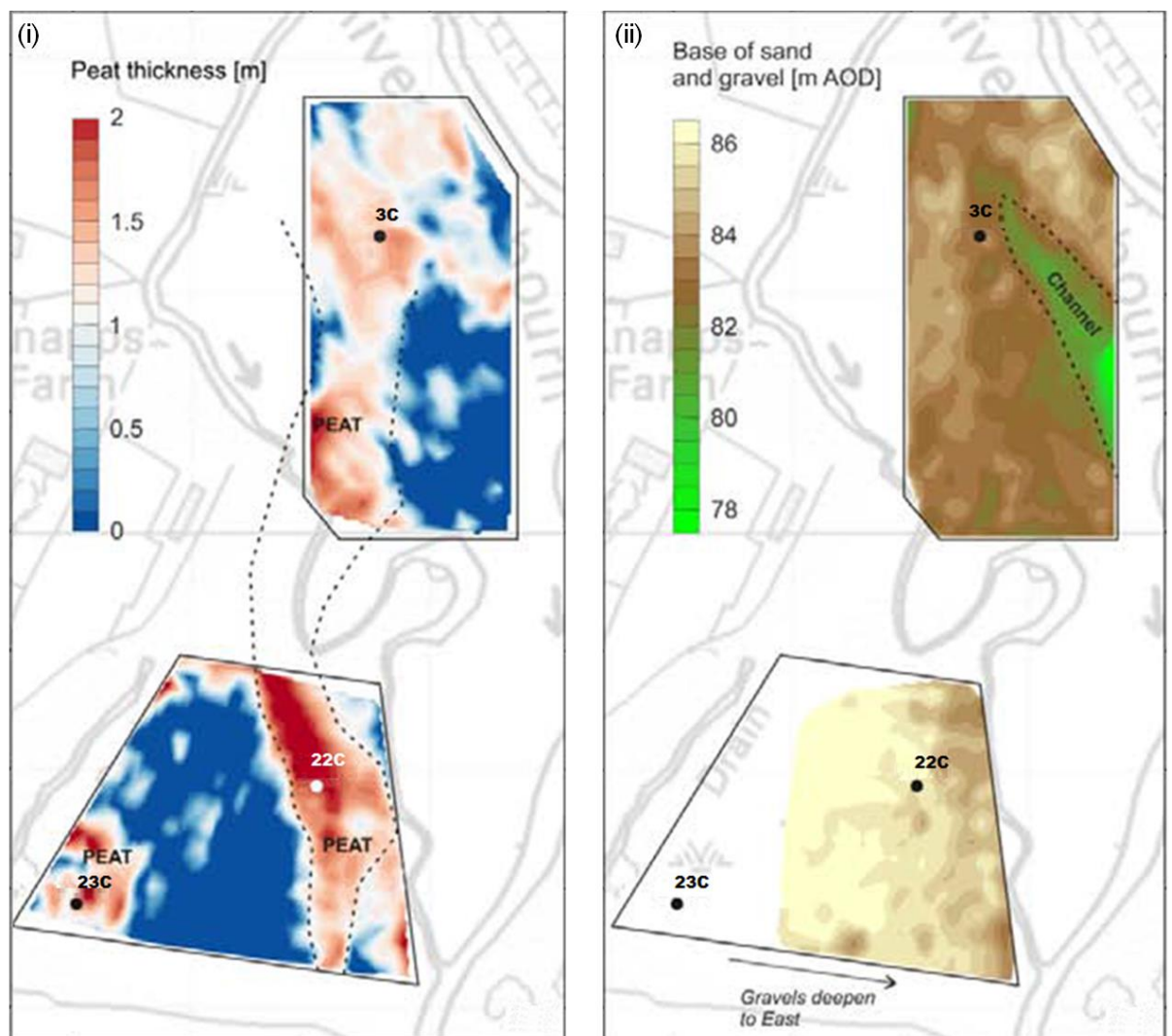


Figure 2-13. Interfaces between geological layers: (i) peat and gravels, and (ii) gravels and Chalk (after Chambers et al., 2014).

interface deepens to the northeast (Figure 2-13ii). This is concordant with deeper scouring where the river bends sharply from east to south. Lateral troughs may denote former channels parallel and subparallel to the existing river course. The most noticeable of these structures is located to the east of the north survey area, running southeast.

2.6 Hydrology

2.6.1 The Lambourn catchment

Draining the Chalk aquifer of the Berkshire Downs, the River Lambourn is characterised by a large baseflow component. At the nearest gauging station, Shaw, 5 km downstream of the Observatory at Newbury (Figure 2-3), the baseflow index and mean discharge are 0.96 and $1.73 \text{ m}^3\text{s}^{-1}$, respectively (Marsh and Hannaford, 2008). The baseflow dominated flow regime is non-flashy, as indicated by the low gradient flow duration curve and percentage exceedance flows (Figure 2-14). Above the 5-10% exceedance value the gradient of the flow duration curve steepens, indicating the infrequent influence of large rainfall-runoff events on the flood regime. Rainfall for the catchment at Shaw was obtained from the CEH-GEAR dataset (Keller et al., 2015), which provides 1 km gridded estimates of daily and monthly rainfall for Great Britain and Northern Ireland derived from the Met Office national database of observed precipitation. Mean annual rainfall for the period 1961-2012 was 747 mm, and mean monthly rainfall 62 mm. Mean monthly rainfall exhibits seasonality, with wetter winters and drier summers (Figure 2-15). Highest mean monthly precipitation occurs in December (77 mm) and lowest in February (51 mm).

In the Lambourn catchment, groundwater in the underlying Chalk aquifer flows in a southerly or south-easterly direction, transverse to the Lambourn at the LOCAR site (Wheater et al., 2006). Transmissivity of the Chalk aquifer increases from $50 \text{ m}^2\text{day}^{-1}$ under interfluvial areas to around $2000 \text{ m}^2\text{day}^{-1}$ in valley bottoms (Allen et al., 1997). Storage coefficients of 0.005 occur under interfluvial areas and 0.015-0.03 in valleys. The alluvial gravels account for a substantial down-valley component of groundwater flow with diffuse vertical water flux, forming an aquifer in themselves (Grapes et al., 2006). These gravels are on the whole highly permeable with variable hydraulic connection to the Chalk (Allen et al., 2010), especially at the river corridor scale (Abesser et al., 2008). Hydraulic conductivity of the gravels has been estimated at ranging from 200 to 7800 md^{-1} (Allen et al., 2010). Poor connectivity between the gravels and Chalk is conceivably due to the presence of heavily weathered chalk beneath the gravels, although lenses of alluvial sands could provide localised connectivity (Wheater et al., 2006). Monthly monitoring of river flow, groundwater levels and water chemistry suggested groundwater discharge

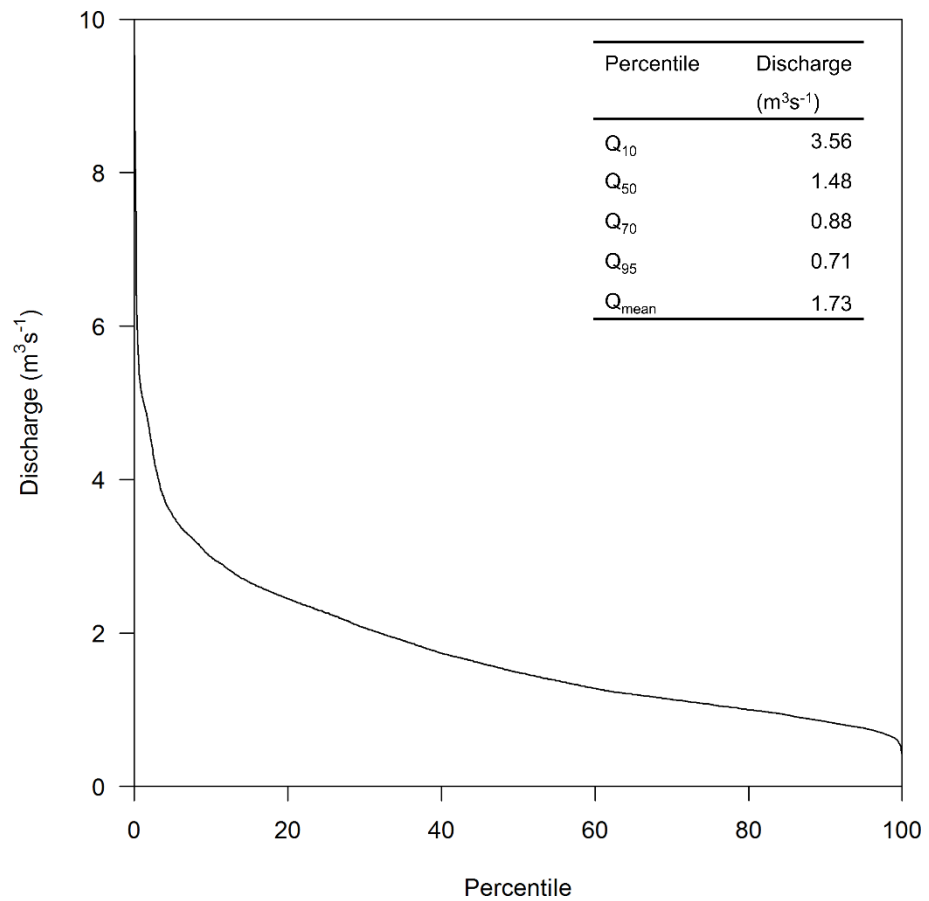


Figure 2-14. Flow duration curve and percentage exceedance flows for the River Lambourn at Shaw (after Marsh and Hannaford, 2008)

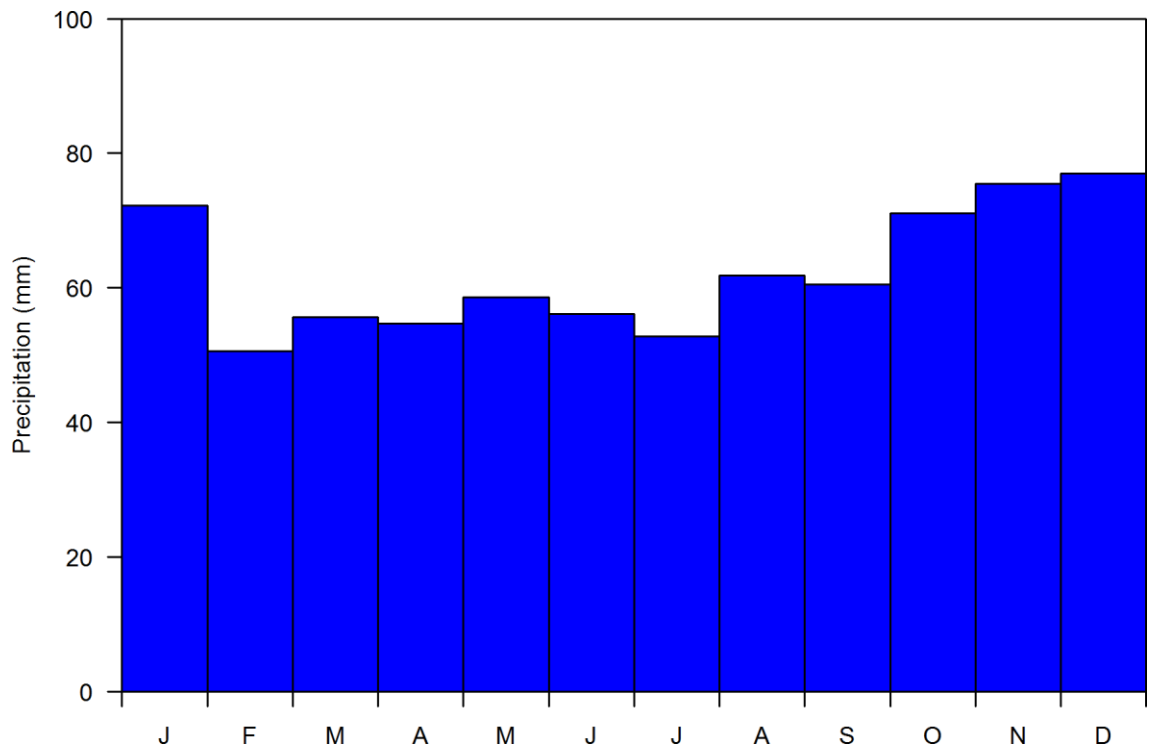


Figure 2-15. Mean monthly precipitation for the Lambourn catchment at Shaw

from the Chalk into the gravels, although significant exchange was shown to be unlikely due to a consistent difference in water levels (Abesser et al., 2008).

The relationship between the river and underlying alluvium involves components of groundwater flow both parallel and transverse to the river, and with both influent and effluent behaviour (Allen et al., 2010). Whilst the gravel aquifer is significant in controlling groundwater-surface water interaction, its importance as a route for flow down the catchment is likely to be modest compared with river discharge. The shallow hyporheic zone, 0-0.5 m below the channel bed, was shown to be highly dynamic as a result of changing surface water inputs from upstream processes (Lapworth et al., 2009). A significant proportion of particulate and colloidal fluorescent organic matter found in the river system and at depth within the underlying gravels indicated surface water inputs were dampened by mixing with deeper groundwater. Groundwater inputs and outputs along the river have complex spatial relationships at a range of scales.

In a small wetland area to the east of the Observatory (Figure 2-16), ground penetrating radar and ERT were used to infer localised areas of groundwater recharge from suppressed seasonal variation in temperature (Musgrave and Binley, 2011). However, due to the study being limited to a single 2D vertical profile, it was not able to establish whether these areas were from lateral flow through the gravel layer or upwelling from the Chalk.

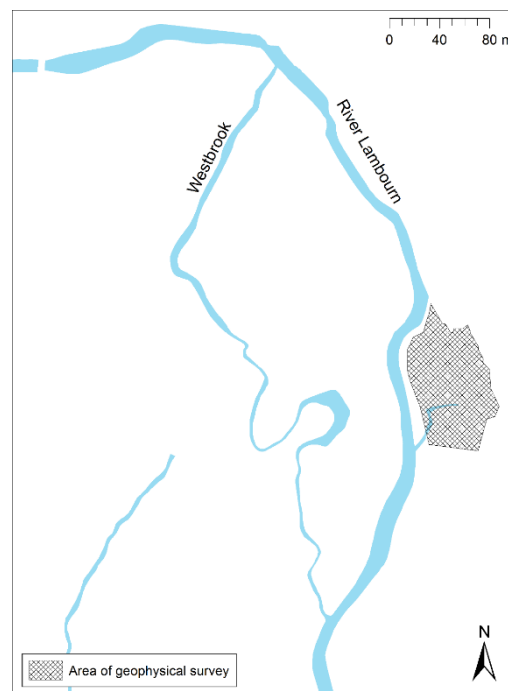


Figure 2-16. Location of eastern wetland site for the Musgrave and Binley (2011) geophysical survey

2.6.2 Investigations within the River Lambourn Observatory

Variable hydraulic relationships across the north and south wetlands were revealed through monitoring for the Habitats Directive (EEC, 1992) between February 2003 and October 2005 (Atkins, 2005; Musgrave, 2006). An investigation based on a piezometer transect and cluster in each meadow (Figure 2-17) indicated good hydraulic continuity between the peat (P) and gravels (G). This was supported by similarities in seasonal changes and an equivalent response to rainfall events at all paired piezometers. Although raw data was unavailable, Figure 2-18, extracted from the Musgrave (2006) study shows that, along the north transect, water levels in the peat were above gravel levels throughout the monitoring period. In the south, peat water levels were below those

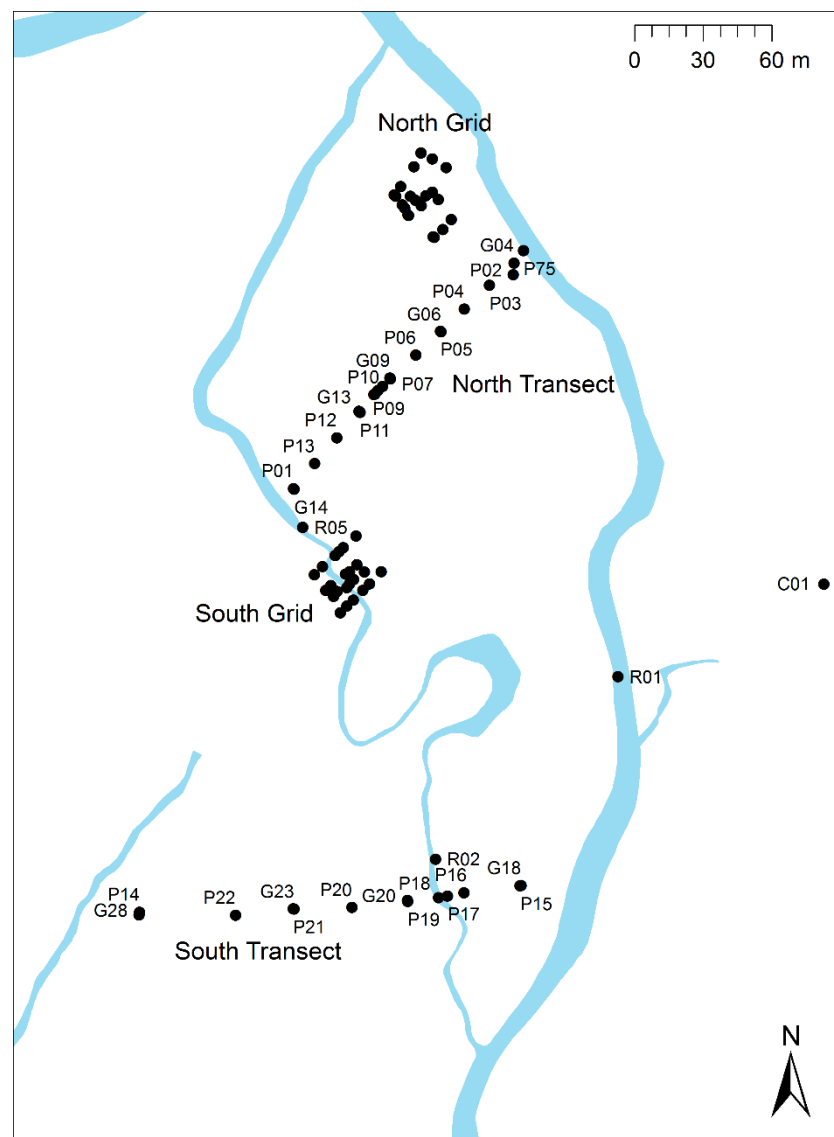


Figure 2-17. Locations of piezometers and stage recorders for the Atkins (2005) and Musgrave (2006) investigations. C, Chalk; G, Gravel; P, Peat; R, Stage

in the gravel. However, head differences were small, in the region of 1-3 cm, and following rainfall, they reversed. Water levels in the Chalk (C01) were consistently above those in both the north and south meadows, with the exception of locations to the east, closer to the Lambourn (P01/13).

The river has been construed as an important control on water levels in the wetland, especially in the north meadow between the Lambourn and Westbrook channels (Musgrave, 2006; Old et al., 2014). An east to west decreasing hydraulic gradient occurred along the north transect during the 2003-2005 monitoring period, with water levels higher in the Lambourn (R01) than in the Westbrook (R05) (Musgrave, 2006) (Figure 2-18). In the south meadow, interactions with the Westbrook channel were not deemed as significant with flow appearing more north to south. A later study in 2009-

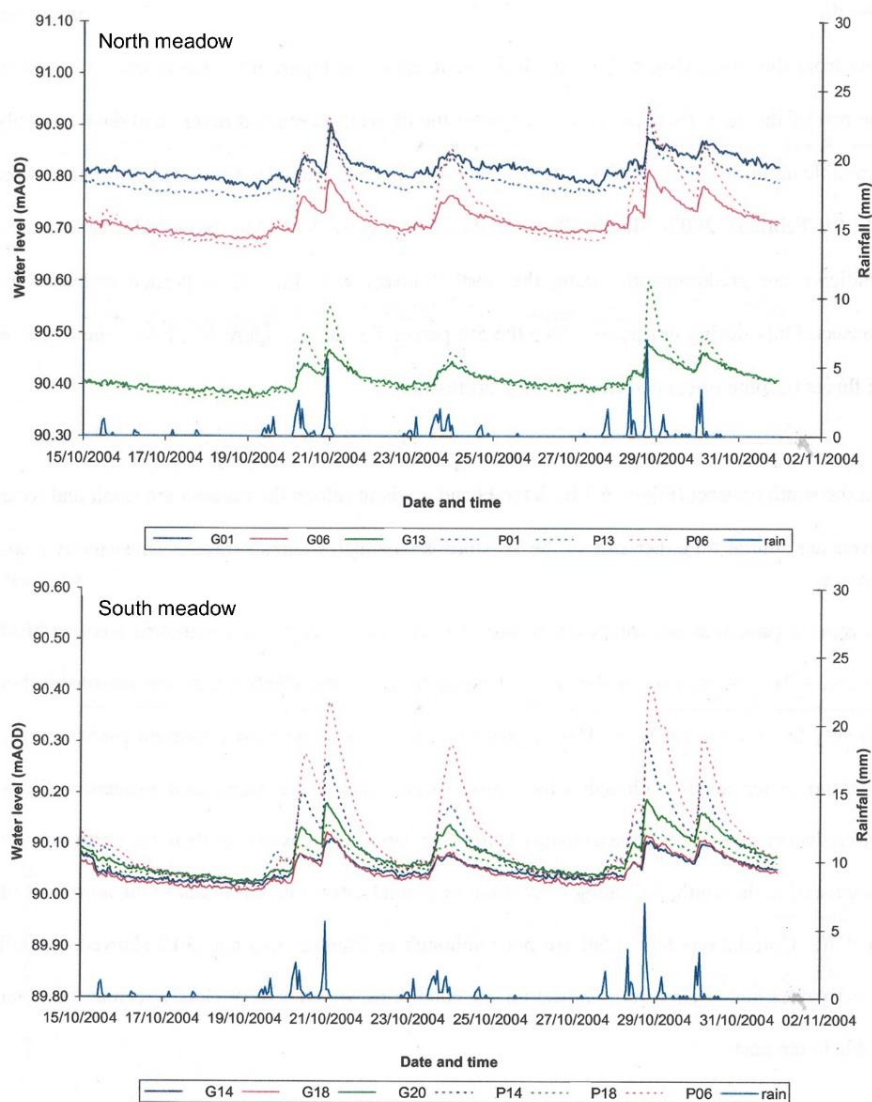


Figure 2-18. Effect of rainfall events on water levels in north and south meadows (Musgrave, 2006)

2010 showed a drop in groundwater levels along the north transect in response to lower Lambourn stage following cutting of instream vegetation (Old et al., 2014) (Figure 2-19).

A calculated monthly water balance for the north meadow resulted in much higher outputs than inputs, with groundwater inflows predominant (Table 2-1) (Musgrave, 2006). In order to have equalised the water balance, net groundwater input would have needed to be halved, either through reducing the vertical head gradient, or hydraulic conductivity. It was construed that heterogeneity in the gravel and peat layers significantly influences the interactions of groundwater with the surface, which was not adequately represented by the monitoring network. This is in line with Chambers et al. (2014), who also pointed to the uneven distribution of the low permeability ‘putty’ chalk acting as a confining layer to the Chalk aquifer, and its implications for exchange between groundwater and surface water. Nevertheless, although upwelling of groundwater to the site provides a significant bulk contribution, the river was generally understood to be the dominant control on wetland water levels, particularly in the north meadow.

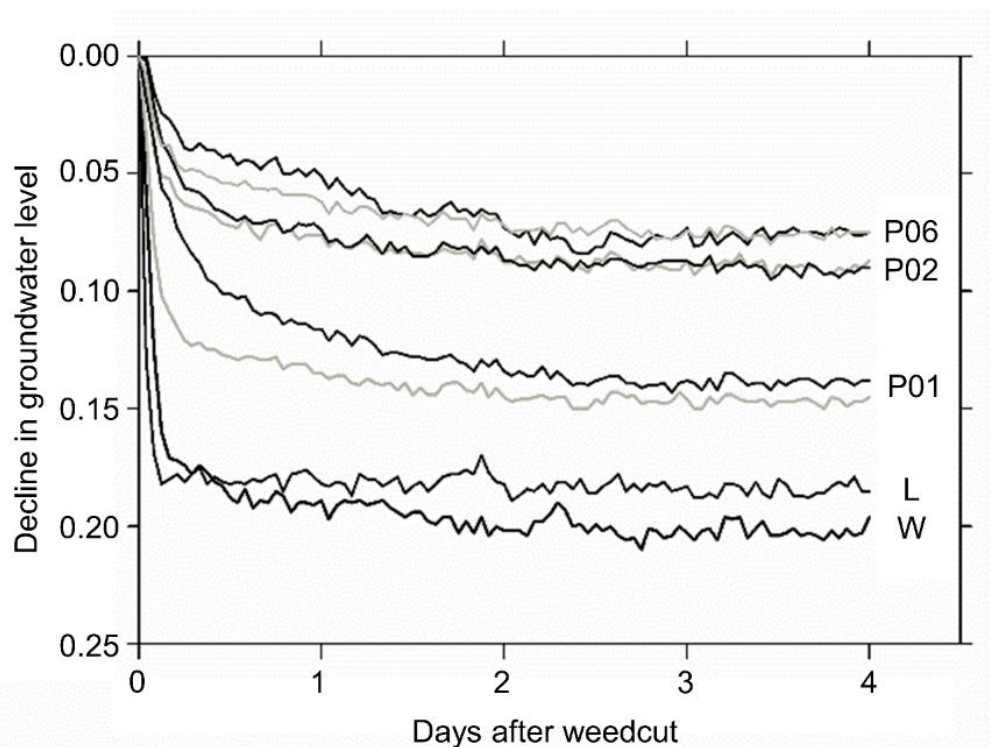


Figure 2-19. Water level responses following a 2010 weed cut in the Lambourn. Peat (black) and gravel (grey) data from paired piezometers at P01, P02 and P06. Lambourn (L) and Westbrook (W) data from stage readings in line with north transect (after Old et al., 2014).

Table 2-1. Monthly and annual water balance for the Observatory north meadow (after Musgrave, 2006). *P*, precipitation; *ET*, evapotranspiration; *L*, lateral flow; *G*, vertical groundwater flow; *R*, channel flow. All values in mm

Component	J	F	M	A	M	J	J	A	S	O	N	D	Annual
P	77	32	51	56	37	38	62	45	36	80	71	64	650
ET	-9	-15	-31	-52	-73	-84	-87	-68	-55	-26	-8	-4	-511
net L	0	0	0	0	0	0	0	0	0	0	0	0	0
net G	312	428	485	482	549	577	489	387	408	166	258	353	4895
net R	-100	-177	-228	-384	-435	-591	-201	-70	59	6	-72	-73	-2265
Total	281	267	276	103	78	-59	263	295	448	226	249	340	2773

2.7 Summary of existing understanding and identification of knowledge gaps

The CEH River Lambourn Observatory comprises a section of the River Lambourn and its riparian wetland floodplain in the Chalk lowlands of southeast UK. Under the Acreman and Miller (2006) classification (see Sections 1.2.1 and 1.3.1.2), the site may be termed a valley bottom groundwater-fed wetland, although the precise nature of the hydrological regime remains ambiguous. Understanding of this is essential in view of the declining presence of Desmoulin's whorl snail and the MG8 community for which the site is designated as SSSI and SAC (Natural England, 2012), yet also the potential effects of the current management regime in periodically cutting back macrophyte growth (Old et al., 2014).

Wetland water levels have been inferred to be principally controlled by the River Lambourn and Westbrook stage (Atkins, 2005; Musgrave, 2006; Old et al., 2014) (Figure 2-20). Groundwater contributions are regarded as more significant in the south meadow, although the possibility of groundwater upwellings at the northernmost extent of the site has been suggested from hydrochemical analysis (Musgrave, 2006). Such deductions were, however, based on an arbitrarily positioned transect and cluster of randomly located piezometers in each meadow.

The geological investigation has been fairly comprehensive, illuminating a complex subsurface architecture of bedrock Chalk, overlying gravels, peat and 'putty' chalk. The low permeability putty chalk is considered to act as a confining layer to the Chalk aquifer, and its uneven distribution has implications for exchange between groundwater and surface water (Chambers et al., 2014). Studies at a nearby site 100 m upstream of the Observatory, part of the LOCAR programme (Wheater et al., 2007), have indicated variable hydraulic connection between the Chalk, gravels and surface water

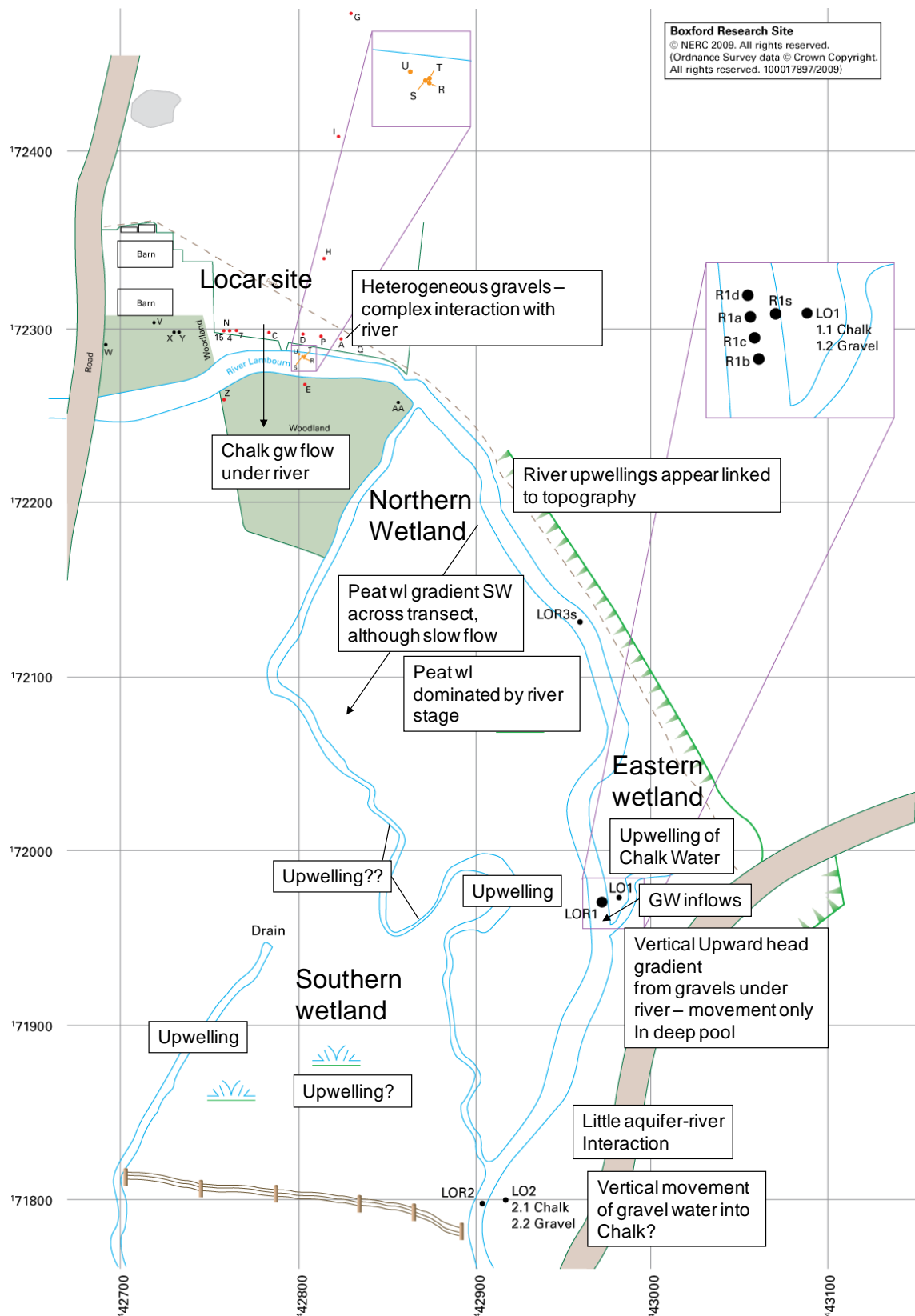


Figure 2-20. Spatial summary of the flow system at the Observatory and its surrounding area according to Allen et al. (Unpublished).

(Abesser et al., 2008; Allen et al., 2010; Lapworth et al., 2009). The extent of this interaction is, however, unclear.

There are a number of areas where the knowledge of the hydrological functioning River Lambourn Observatory may be improved. The uncertainty around the degree of groundwater/surface-water interaction warrants further examination. A more spatially comprehensive treatment of groundwater and surface water levels is required in the wetland. Towards this end, the spatial extent of the putty chalk layer and its absences, along with possible areas of groundwater upwelling, need delineating. The River Lambourn discharge, water levels and channel geometry are essential, as is an expansion of the current topographical extent to include the entire site. Meteorological data, in the form of precipitation and evapotranspiration, necessary to the water balance for the site, need to be updated. Care of these gaps in understanding enables the development of a robust conceptual model, and forms the basis of the next chapter.

Chapter 3

Developing a conceptual model of hydrological functioning

3.1 Introduction

Despite previous research into the hydrology and geology of the River Lambourn Observatory, and a nearby site 100 m upstream (see Chapter 2), water fluxes at the site remain poorly understood. To address knowledge gaps an extensive and targeted field campaign was implemented. A primary concern was the implications for hydraulic continuity between different subsurface layers and surface waters caused by complex geological variations. To elucidate the degree and mechanisms of groundwater/surface water interaction surveys of subsurface temperature, hydrochemistry and vegetation were carried out in conjunction with monitoring of groundwater levels and channel stages. The spatial extent of the highly weathered putty chalk was extracted from the existing ERT survey data. Comprehensive site characterisation also necessitated investigation into the fluvial regime of the River Lambourn and Westbrook, including channel cross-section surveys and measurement of flows. Meteorological observations were recorded after installation of an automatic weather station, whilst the existing topographical survey was extended to incorporate areas overlooked. The field campaign and subsequent analysis, detailed in this chapter, enabled the development of a conceptual model of hydrological functioning for the Observatory.

3.2 Site instrumentation and scheduled monitoring

Monitoring at the Observatory took place from 1/2/2013 until 1/10/2014, to coincide at start with full installation of meteorological equipment and end with decommissioning of groundwater monitoring equipment. The full instrumentation network is detailed in Figure 3-1.

Continuous 15 minute averaged meteorological observations were logged using an automatic weather station (AWS) (Figure 3-2i). Air temperature and relative humidity were recorded using a CS215TM sensor. An R M Young 03101TM cup anemometer measured wind speed at 2 m above ground level. Subsurface temperature at 0.1, 0.3 and 0.5 m depths were recorded with 107-LCTM temperature sensors. Solar radiation was recorded with a LP02TM pyranometer. Accumulated 15 minute precipitation was measured with an ARG100TM tipping bucket rain gauge (Figure 3-2ii).

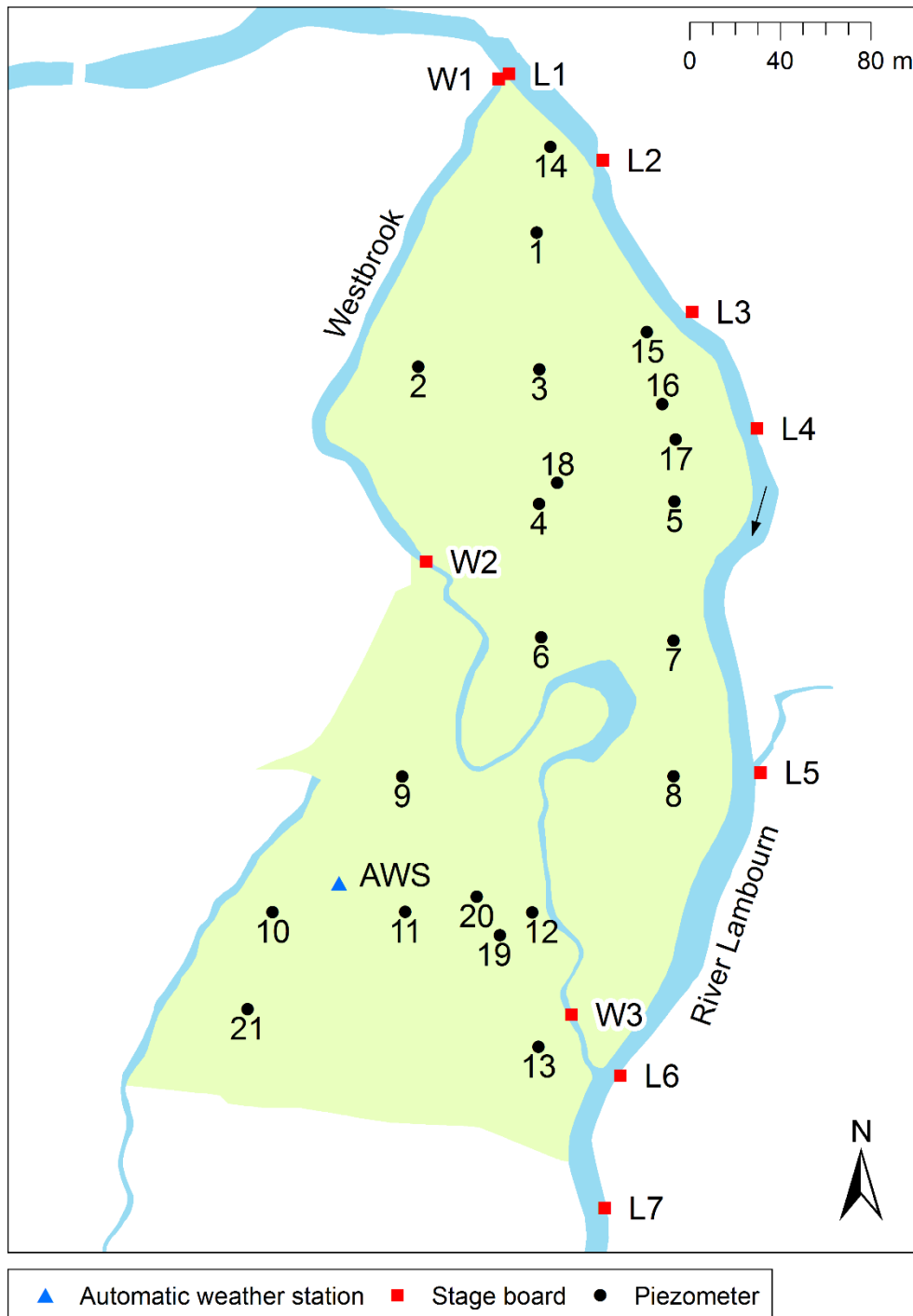


Figure 3-1. The instrumentation network for the Observatory

An existing gridded piezometer array originally installed by CEH in February 2012 was numbered 1-13. Supplemental piezometers to the same design specification were added in May 2013 to target observed temperature anomalies (locations 14-19). All locations comprise separate peat (P) and gravel (G) piezometers (Figure 3-2iii), with the exception of location 8 where the peat was too thin to complete an installation. Gravel piezometers were screened approximately 2.5-3.5 m bgl while peat piezometers were screened across the entire peat thickness. Peat piezometers in the pre-existing array were



Figure 3-2. Photos of (i) the Automatic Weather Station, (ii) tipping bucket rain gauge, and (iii) piezometer pair

installed with the slotted screen extending above ground level, while bentonite was used to seal new piezometers with closed screens above ground level. Chalk (C) boreholes are also located at sites 3, 20 and 21; these were screened at 9.5-10.0, 8.0-9.0 and 5.0-6.0 m bgl, respectively.

Groundwater heads are routinely checked at all piezometers by manually dipping observed water levels. At peat piezometer locations 3-6, 11-15 and 17, and gravel piezometer locations 3-6, 11, 12, 14, 15, 18 and 19, groundwater heads were monitored every five minutes using either In-Situ Level Troll 500s™ or SWS Divers™ installed to a consistent depth of 3 m bgl in gravel piezometers and to the base of the peat in peat piezometers. Channel stage was observed monthly at seven stage boards along the River Lambourn (L1, L3-L7) and three in the Westbrook (W1-W3). River Lambourn stage was also recorded every 15 minutes at L2 using a SWS Diver™ installed in a stilling well. Additionally, gravel groundwater temperatures are monitored every five minutes using the same In-Situ Level Troll 500s™ or SWS Divers™ installed for groundwater heads. The River Lambourn temperature is measured every 15 minutes at L2, with a Druck PDCR 1830™.

3.3 Extension of topographical coverage

The existing topographic survey area (Section 2.4) was extended in April 2014 to incorporate omitted regions that had become accessible and capture the drainage channel to the southwest. An additional 1195 points, located in the south and also

towards the west boundary, were georeferenced with Trimble R8™ dGPS at noticeable changes in slope (Figure 3-3).

Poor accessibility and connectivity to the satellite network in overgrown areas limited the overall survey area. In order to extend coverage to the full site 1 m resolution LIDAR data was obtained from the Environment Agency's Geomatics Group for Ordnance Survey 1 km grid squares SU4271, SU4272, SU4371 and SU4372. Ground survey and LIDAR data sets were combined to produce a comprehensive digital elevation model (DEM) of the Observatory and surrounding area (Figure 3-4) The combination of datasets was achieved through use of the mosaic function with the mean operator in ESRI ArcGIS™, where the output raster cell value of overlapping rasters is the average of the input cells. Elevations were then interpolated via a two-dimensional minimum curvature spline technique.

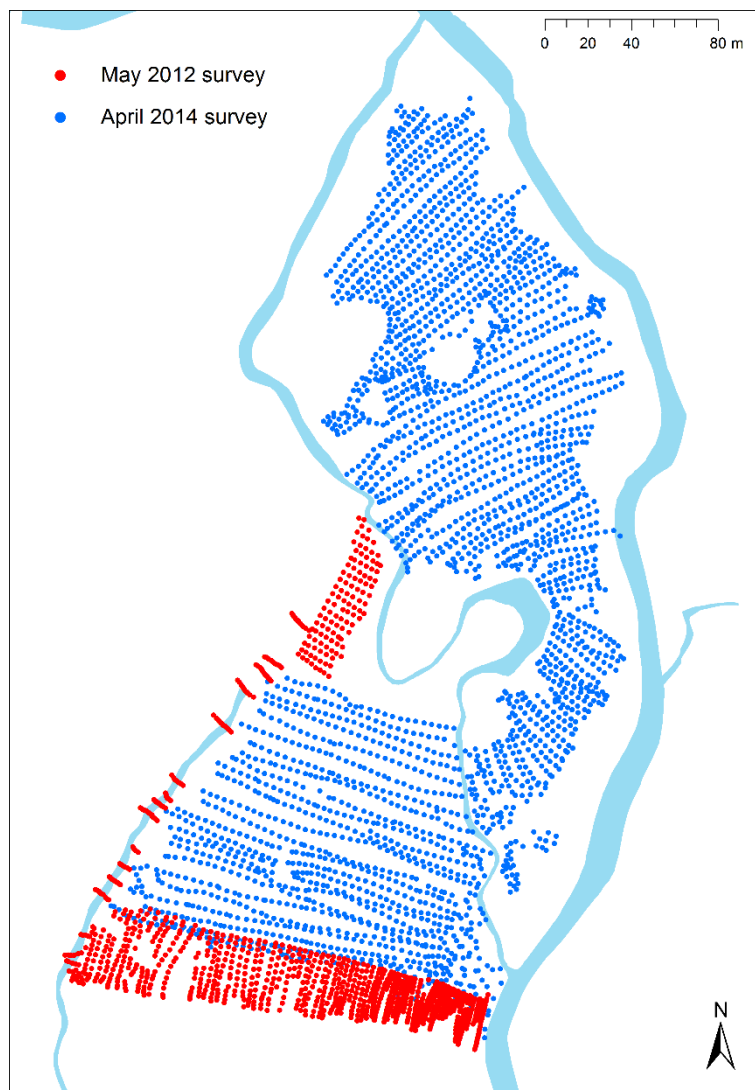


Figure 3-3. Locations for May 2012 and April 2014 topographic surveys.

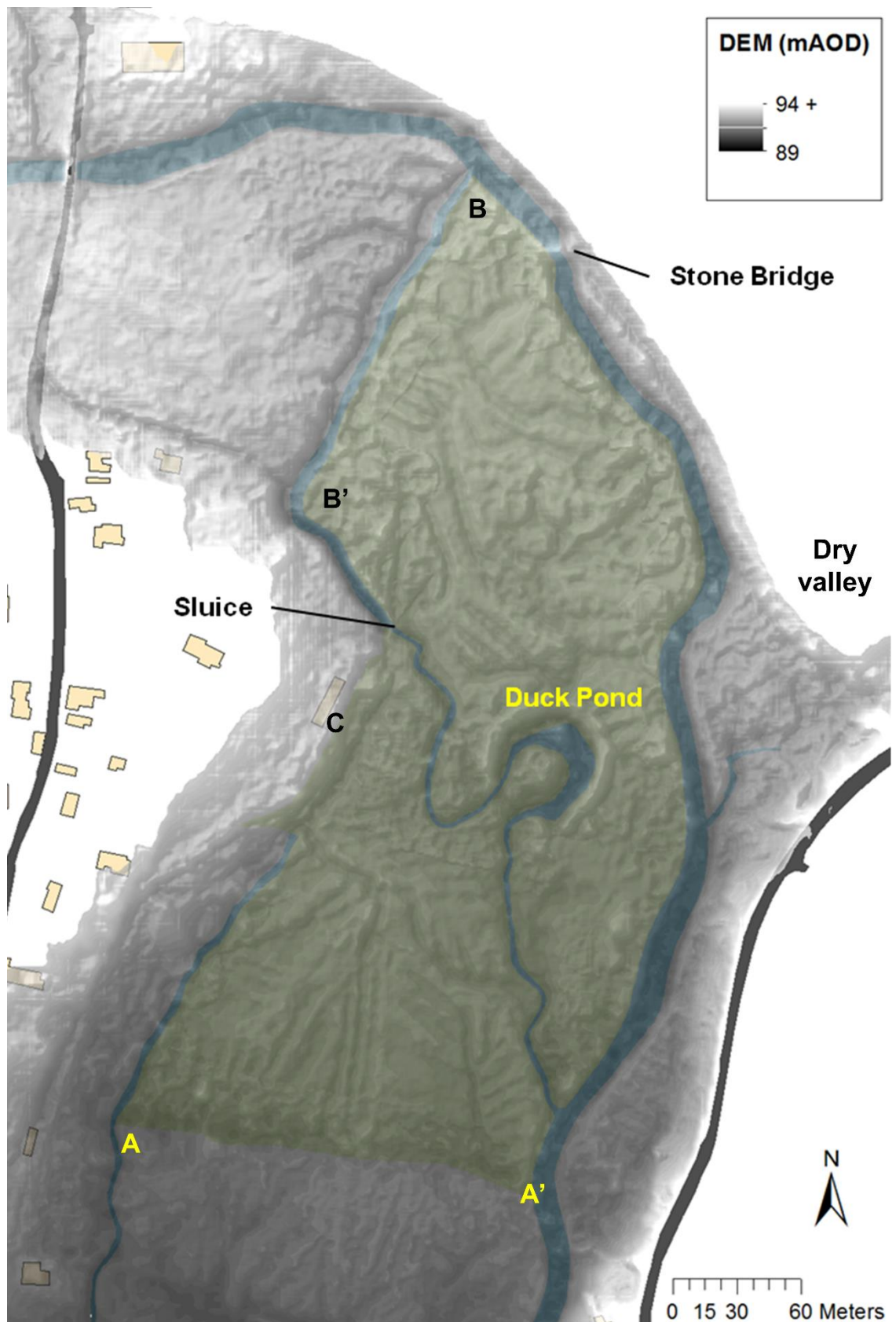


Figure 3-4. DEM of Observatory and Lambourn valley from topographic surveys and LIDAR, showing channel structure and modification locations.

Increased coverage showed the elevation of the Observatory to range from 89.43 to 97.51 mAOD, with mean elevation at 90.70 mAOD. There is a gentle slope from northwest to south, following the line of the valley bottom, of 0.016. The dendritic channel network which crosses the peat, following the gradient, is conspicuous. Also noticeable are the excavated duck pond in the centre of the site, and the dry valley to the east.

A number of features outside the boundary of the pre-existing survey are revealed by the extended topographic survey. These are mostly part of the surface drainage network and include a wide depression running along the southern border of the site between the River Lambourn (A - A', Figure 3-4) and the southwest drainage channel. The route of the Westbrook is defined, with a disused supply channel running parallel in the north meadow (B - B', Figure 3-4). A supply channel may also be observed running from the sluice into the south meadow (C, Figure 3-4), where it splits into the southwest drainage channel and a course into the centre of the south meadow.

3.4 Meteorological observations

3.4.1 Precipitation

Total measured precipitation recorded by the AWS for the full monitoring period was 1471.4 mm, while for the complete hydrological year (1/10/2013-30/9/2014) of the monitoring period it was 1081.4 mm, well above the mean annual rainfall for the Lambourn catchment at Shaw (747 mm, Section 2.6.1). Short periods of noticeably intense precipitation occurred on 18/7/2014 (7.2 mm day⁻¹), 19/9/2014 (6.6 mm day⁻¹) and 20/10/2013 (6.4 mm day⁻¹) (Figure 3-5). The highest monthly precipitation was 211.0 mm in January 2014, with the monthly average at 73.6 mm. Periods of sustained precipitation occurred in October 2013 and December 2013 to mid-February 2014, contributing to high monthly totals over these periods. A distinct dry spell is conspicuous in March 2014 with a monthly total of 22 mm, while smaller dry periods occurred through March to September 2013 and in September 2014. The period of heavy precipitation over the winter of 2013/2014 is likewise conspicuous. Some seasonality is apparent, though not obvious.

3.4.2 Evapotranspiration

Potential evapotranspiration was calculated using the Penman-Monteith formula (Monteith, 1965; Penman, 1948) from data supplied by the AWS. Subhourly and daily evapotranspiration rates are shown in Figure 3-6, and monthly in Figure 3-7. Total potential evapotranspiration through the full monitoring period was 1390.1 mm, and for

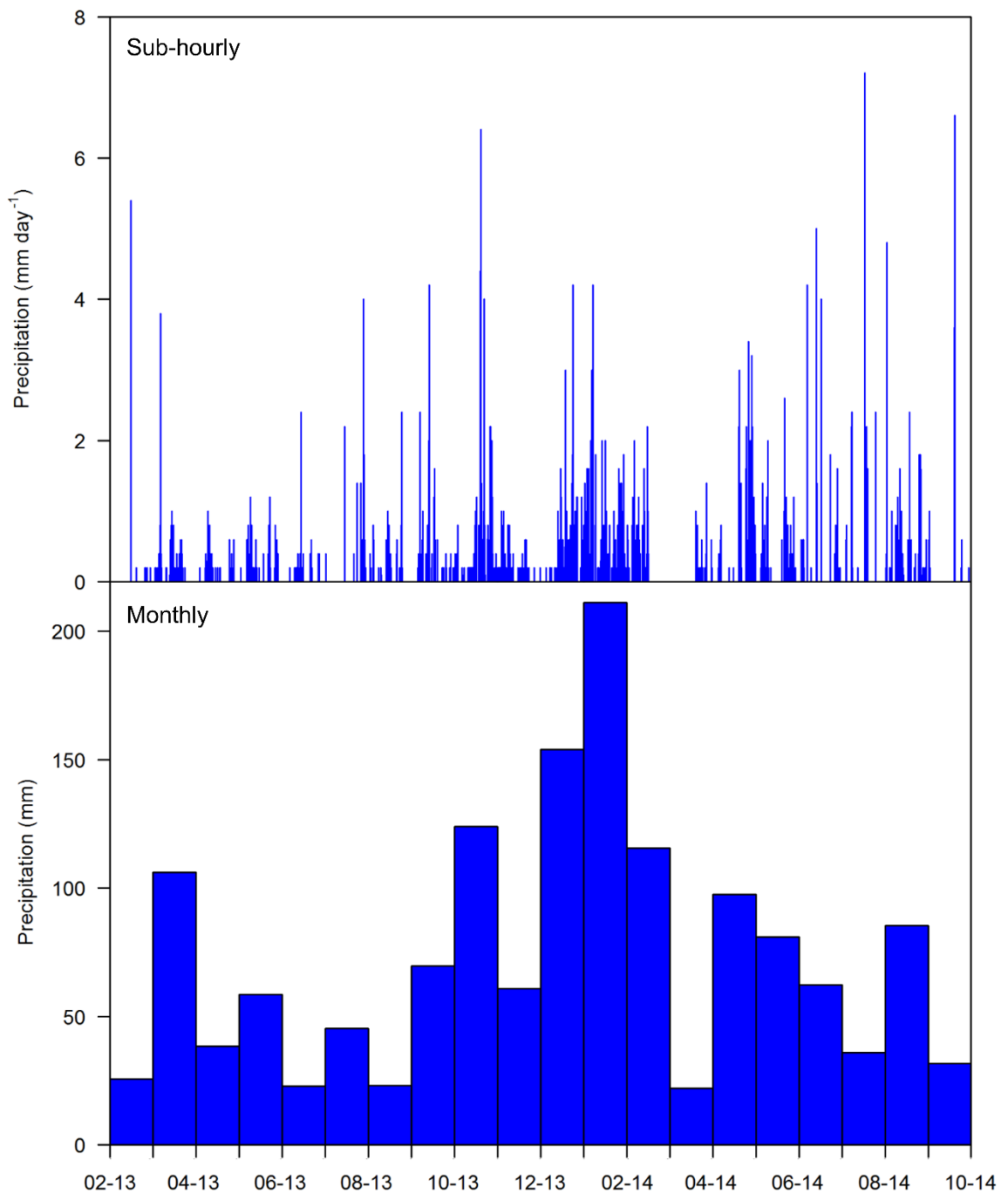


Figure 3-5. Sub-hourly (15 min) and monthly total precipitation for the monitoring period
1/2/2013 – 1/10/2014

the complete hydrological year 764.3 mm. Maximum subhourly potential evapotranspiration for the period was 6.2 mm day^{-1} on 29/3/2014, while maximum daily evapotranspiration occurred on 18/4/2013 at 9.2 mm day^{-1} . The mean subhourly 0.5 mm day^{-1} and daily 2.3 mm day^{-1} rates indicate such high rates of evapotranspiration are not the norm. Potential evapotranspiration minima of 0.0 mm day^{-1} occurred at

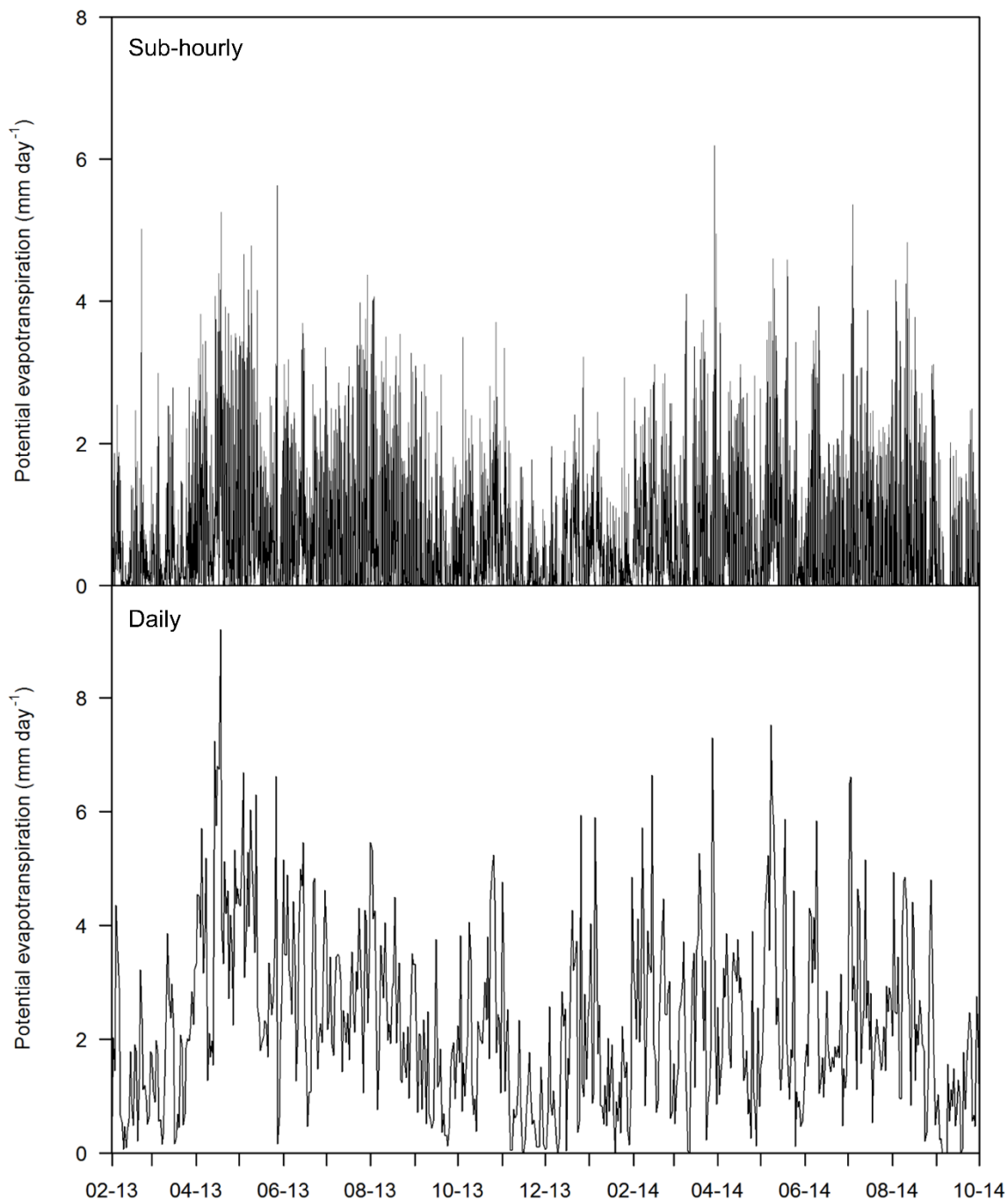


Figure 3-6. Sub-hourly (15 min) and daily total potential evapotranspiration for the monitoring period 1/2/2013 – 1/10/2014

several points throughout the period. Average monthly potential evapotranspiration was 69.5 mm, with the maximum 126.8 mm and minimum 27.0 mm occurring in April 2013 and September 2014, respectively. There were generally greater rates of potential evapotranspiration through the periods May 2013 – August 2013 and March 2014 – August 2014.

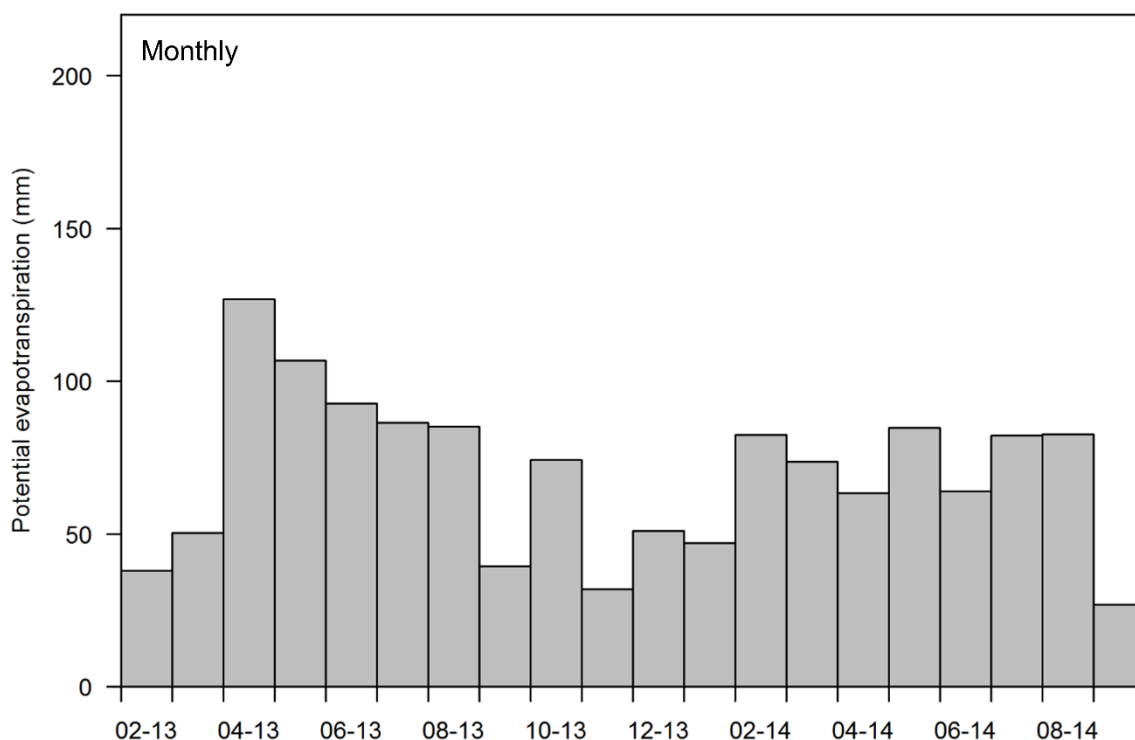


Figure 3-7. Monthly total potential evapotranspiration for the monitoring period 1/2/2013 – 1/10/2014

3.4.3 Net precipitation

Although net precipitation is a function of actual evapotranspiration rather than potential evapotranspiration, the site is likely to be energy-limited with groundwater and runoff dominant. Hence, it is not expected that the difference between actual and potential evapotranspiration is significant. Here, net precipitation is defined as precipitation minus potential evapotranspiration (P-PET), which shows a total difference of 81.3 mm net precipitation for the full monitoring period, yet 317.0 mm for the full hydrological year (October 2013 – September 2014) within this period (Table 3-1). The seasonality to some extent evident in the precipitation and potential evapotranspiration time series is given emphasis. Greatest net precipitation occurred through December 2013 (102.7 mm) and January 2014 (164.0 mm). Negative net precipitation, or net evapotranspiration, was seen in January 2013, through April 2013 – August 2013, March 2014 and May 2014 – July 2014. This peaked at -88.4 mm in April 2013.

3.5 River Lambourn and Westbrook morphology and hydraulics

3.5.1 Channel geometry

The River Lambourn and Westbrook both bear the marks of historical use, with evidence of modifications and additions throughout their length. Two main structures in the

Table 3-1. Monthly total precipitation (P), potential evapotranspiration (PET) and precipitation minus potential evapotranspiration (P-PET)

Month	P (mm)	PET (mm)	P-PET (mm)
02-13	25.6	38.0	-12.4
03-13	106.2	50.4	55.8
04-13	38.4	126.8	-88.4
05-13	58.6	106.7	-48.1
06-13	23.0	92.8	-69.8
07-13	45.4	86.3	-40.9
08-13	23.2	85.2	-62.0
09-13	69.6	39.5	30.1
10-13	124.0	74.2	49.8
11-13	60.8	32.0	28.8
12-13	153.8	51.1	102.7
01-14	211.0	47.0	164.0
02-14	115.6	82.4	33.2
03-14	22.0	73.6	-51.6
04-14	97.6	63.4	34.2
05-14	81.0	84.7	-3.7
06-14	62.4	64.0	-1.6
07-14	36.0	82.3	-46.3
08-14	85.4	82.7	2.7
09-14	31.8	27.0	4.8
Total	1471.4	1390.1	81.3

channels constitute a stone bridge in the River Lambourn (Figure 3-8), and sluice gate in the Westbrook (Figure 3-9), located as shown in Figure 3-3. In order to evaluate channel morphology throughout the study reach and expand on the influence of these structures, channel cross-section surveys were conducted using Trimble R8™ dGPS for the Westbrook in May 2013 and the River Lambourn in November 2013.

Cross-sections were surveyed at 44 locations along the Westbrook and 42 along the River Lambourn (Figure 3-10). Locations were restricted in access and line of sight by dense vegetation. For the purposes of this study, bankfull stage was observed as an abrupt change in slope, dominant bench level and, if applicable, the lower limit of riparian vegetation.

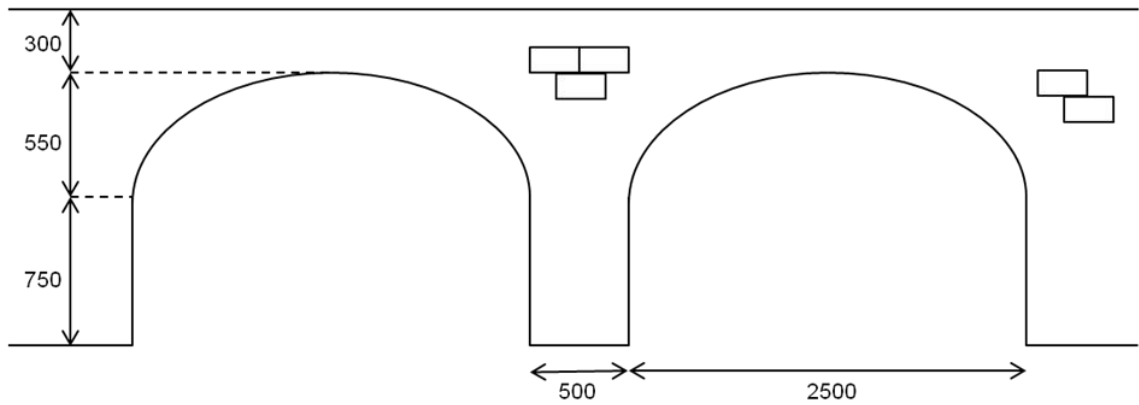


Figure 3-8. Photograph and schematic (dimensions in mm) of the stone bridge over the River Lambourn

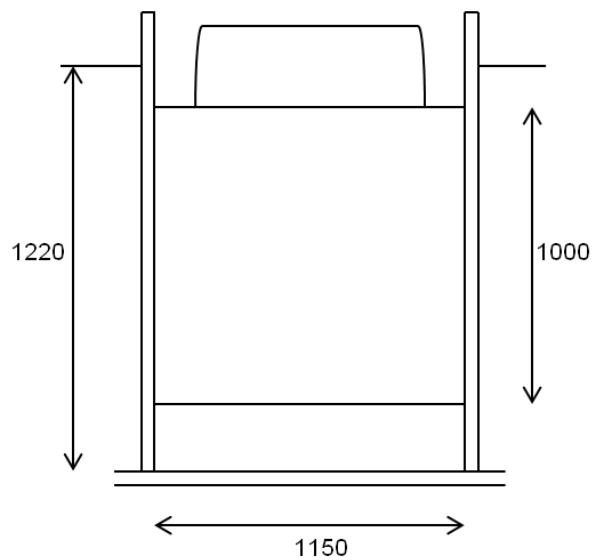


Figure 3-9. Photograph and schematic (dimensions in mm) of the Westbrook channel sluice



Figure 3-10. Locations for River Lambourn (L) and Westbrook (W) channel cross-sections

Selected cross-section profiles at L10, L30, W6, W13, W32 and W41 illustrate the morphological range characteristic for the Observatory (Figure 3-11). The study reach of the River Lambourn, which has an average width 11.7 m, cross-sectional area 9.4 m², depth 1.0 m, and maximum depth 1.8 m, is represented primarily by the cross-section at L10. The longitudinal profile (Figure 3-12) shows the bed to be relatively uniform at a gradient of 0.0025. Depressions at 70 , 235 and 400 m downstream from L1 signify pools. Cross-section L30 coincides with the latter of these. The pool at 70 m is situated immediately downstream of the stone bridge and likely represents the effects of afflux. A noticeable drop in the elevation of the left bank occurs just upstream, where the bridge footings extend to create a backwater.

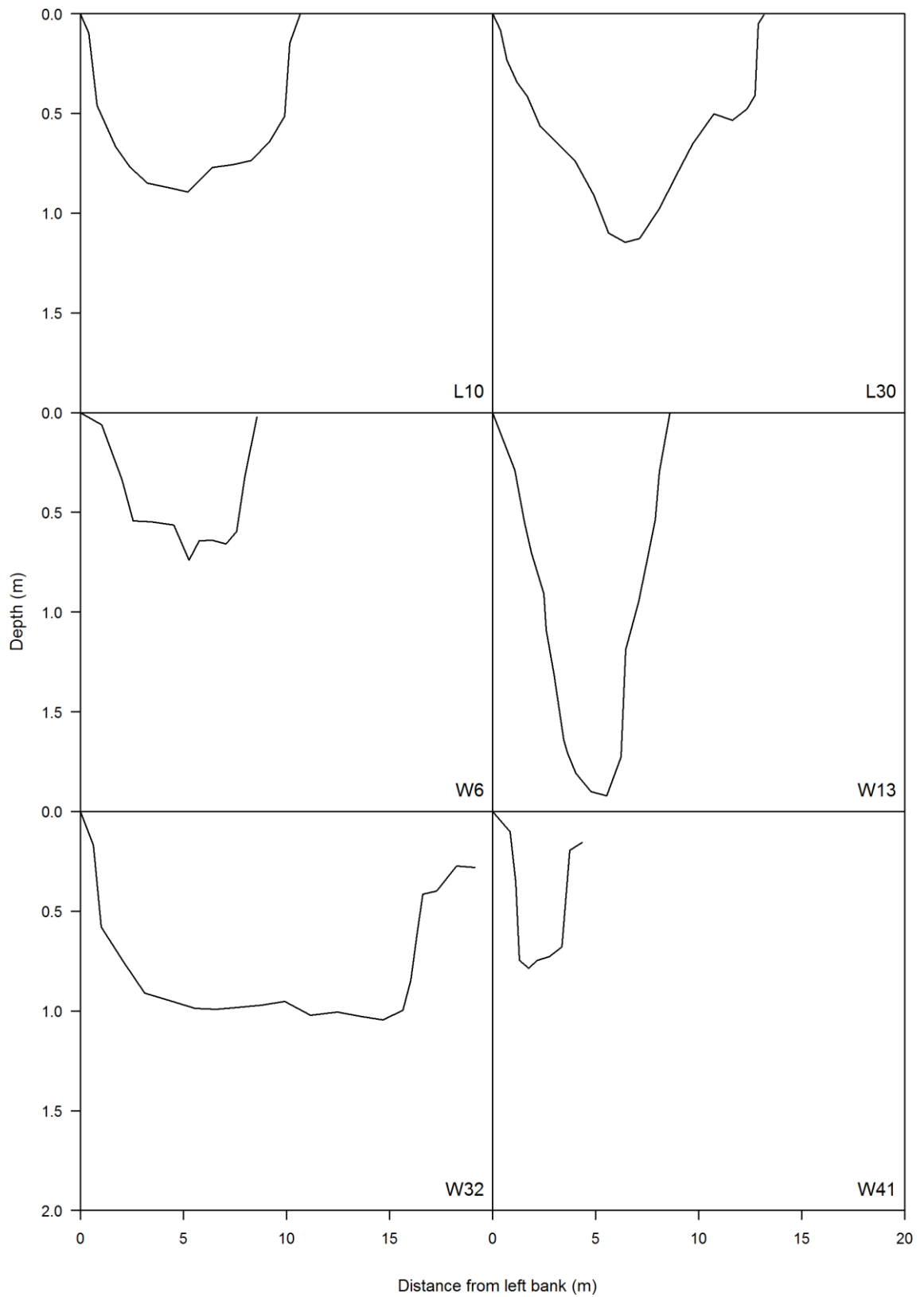


Figure 3-11. Selected channel cross-section profiles for the River Lambourn (L) and Westbrook (W)

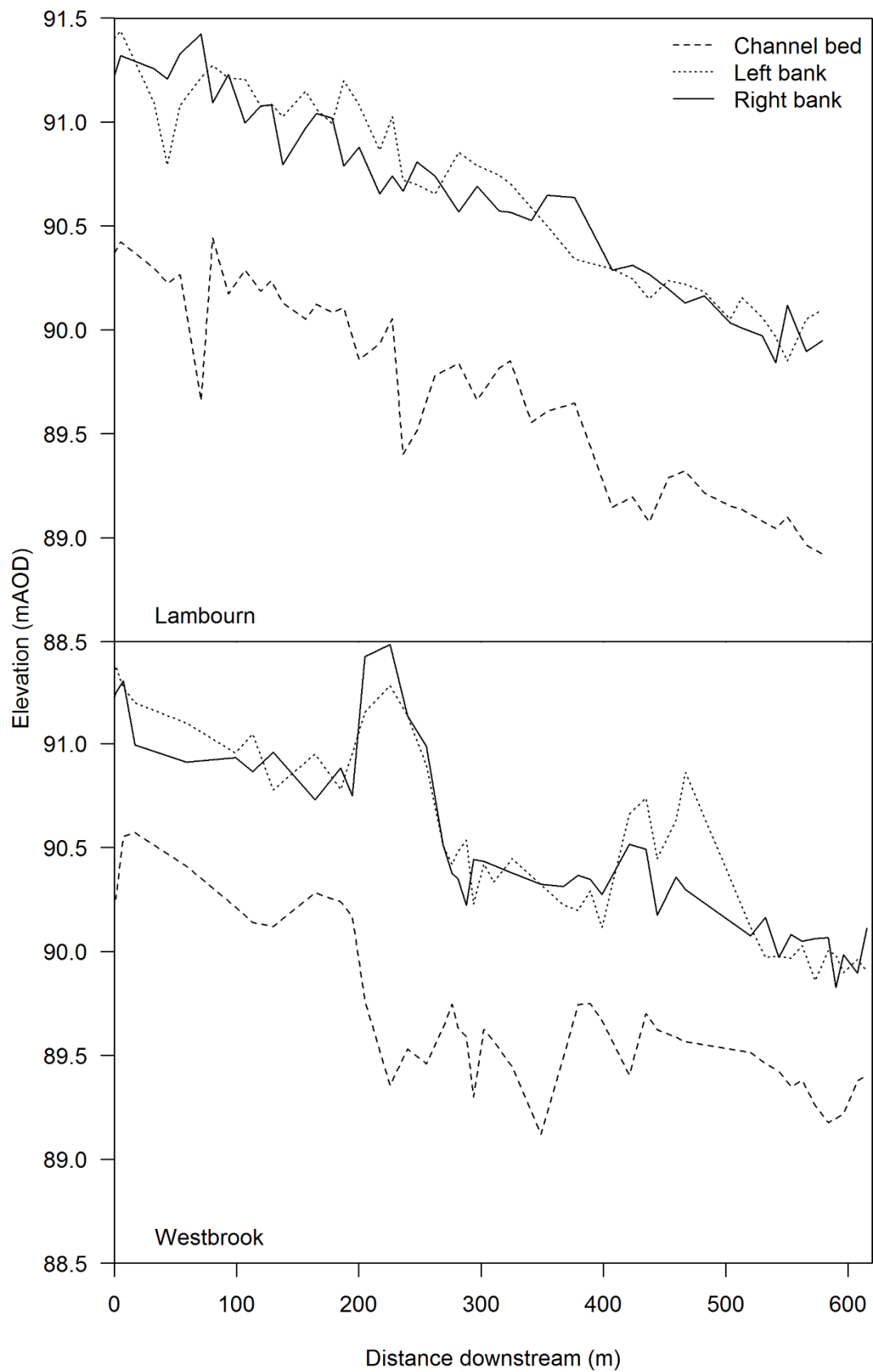


Figure 3-12. Longitudinal profiles of channel bed, left bank and right bank elevations for the River Lambourn and Westbrook

Greater variation in channel morphology is displayed in the Westbrook, mainly due to human modification. The channel, which has an overall gentler slope at 0.0018, may be divided into four sections with distinct channel geometry. Branching off from the River Lambourn at the northwest boundary of the site, a 130 m length straight section is typified by cross-section W6. A sharp bend leads into a section upstream of the sluice gate where the channel has been widened slightly and deepened down to almost 2 m. This artificial pool, represented by W13, extends for approximately 90 m. Material dug out from the bed was deposited on the banks causing the sudden rises in elevation seen in the longitudinal profile. This is also the case for profile W32, which shows the widened channel for the duck pond with excavated material deposited on the banks. Upstream from the duck pond until the sluice gate, and downstream to where the Westbrook re-joins the River Lambourn, the channel becomes shallower and narrows considerably as demonstrated by profile W41.

3.5.2 Measurement of discharge

River Lambourn discharge has been measured monthly at L1 since mid-2009 using an electromagnetic flow meter (Figure 3-13). A strong regression relationship ($R^2 = 0.99$) between these measurements and corresponding flows at the downstream Shaw gauging station (Figures 3-13 and 3-14) was used to derive a 15 minute time series for the monitoring period (Figure 3-15).

A strong seasonal signal is apparent in the derived flow record, with winter highs in February 2013 – May 2013 and January 2014 – May 2014 dropping through May and June 2013 and 2014 to summer lows. Increases from storm events are noticeable, while the sharp increases in January 2014 and February 2014 correspond to the sustained heavy rainfall recorded through the period (Figure 3-5). Mean discharge for the monitoring period was $2.29 \text{ m}^3\text{s}^{-1}$, while the maximum $8.15 \text{ m}^3\text{s}^{-1}$ occurred on 15/2/2014. The minimum discharge $0.66 \text{ m}^3\text{s}^{-1}$ was seen on 10/10/2013.

The low gradient flow duration curve and percentage exceedance flows indicate a non-flashy regime (Figure 3-16). The mean discharge for the monitoring period corresponds to 39.9% exceedance, whilst the magnitude of the flow exceeded 50% of the time was $1.69 \text{ m}^3\text{s}^{-1}$. Flows exceeded 10% of the time were greater than $4.15 \text{ m}^3\text{s}^{-1}$, relating to the period of high flow February 2014 – April 2014. Above the 10% exceedance value the gradient of the flow duration curve clearly steepens, indicating the influence of rainfall-runoff events on the flood regime. Continuous flows throughout the period and the flat gradient at low flows are indicative of base flow sustained by groundwater (Ward and Robinson, 2000).

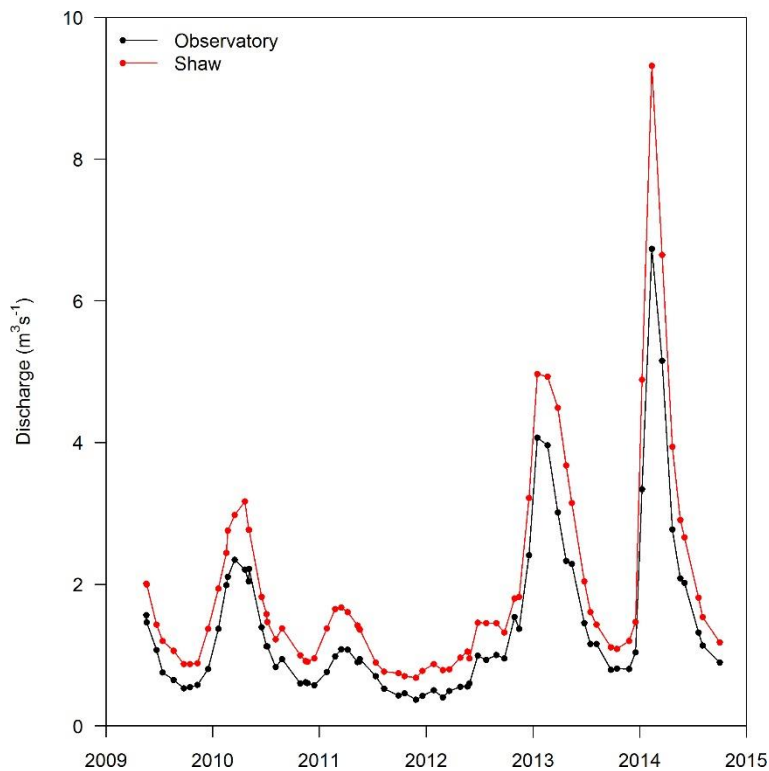


Figure 3-13. Monthly measurements of discharge for the River Lambourn at the Observatory and corresponding discharge at Shaw gauging station 2009 - 2015

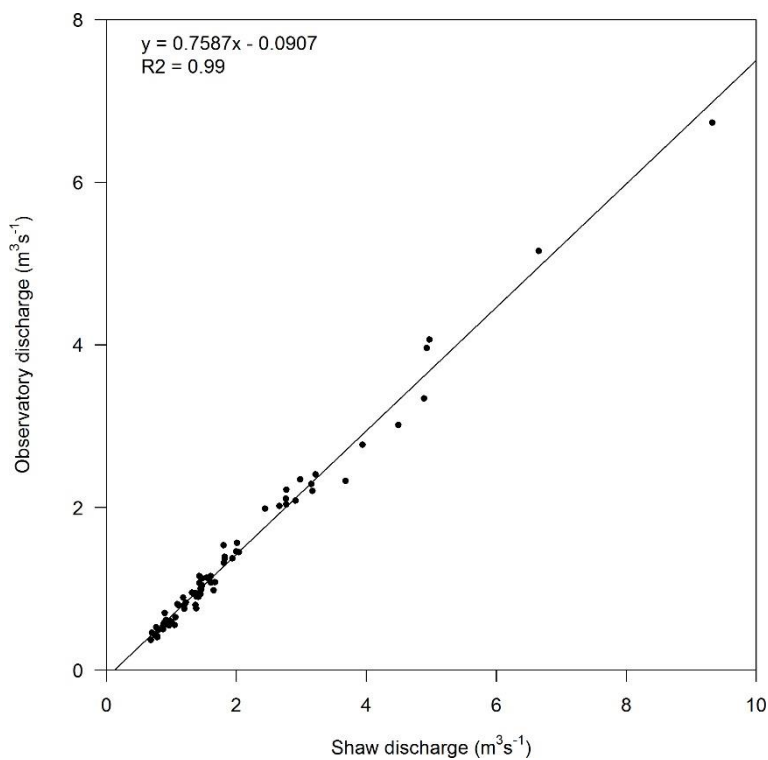


Figure 3-14. Regression relationship between monthly measurements of discharge for the River Lambourn at the Observatory and corresponding discharge at Shaw gauging station

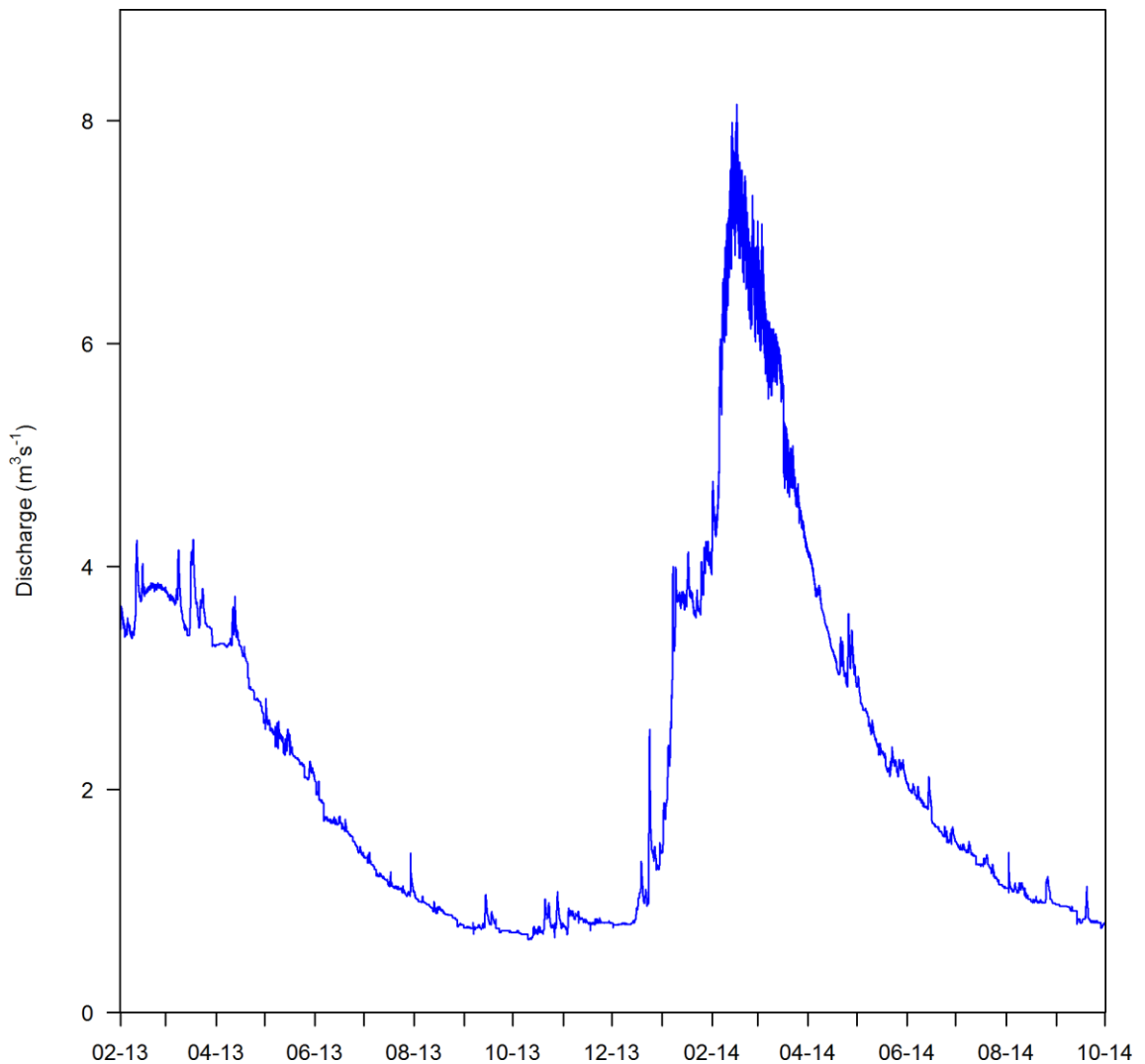


Figure 3-15. Sub-hourly (15 min) derived discharge for the River Lambourn at the Observatory over the monitoring period 1/2/2013 – 1/10/2014

3.5.3 Surface water stage and the influence of macrophytes

River Lambourn monthly stage board readings and 15 minute logged stage show temporal changes in stage were similar at all locations (Figures 3-17 and 3-18). Over the reach downstream changes in water surface elevation reproduce the bed slopes of 0.0025 and 0.0018 in the River Lambourn and Westbrook respectively. Periods of low stage are noticeable mid-July 2013 – January 2014 and July 2014 – October 2014. The high stage levels observed over January 2014 – April/May 2014 echo the high flows and sustained heavy precipitation recorded for the same period. However, in-channel seasonal macrophyte growth and its management through cutting (Figure 3-19) dominate changes in water levels. Weed cuts on 1/5/2013, 16/7/2013, 21/5/2014 and 23/7/2014 signify rapid decreases in stage which otherwise fluctuates in response to the growing season. Macrophyte growth is a principal influence on fluvial dynamics in a

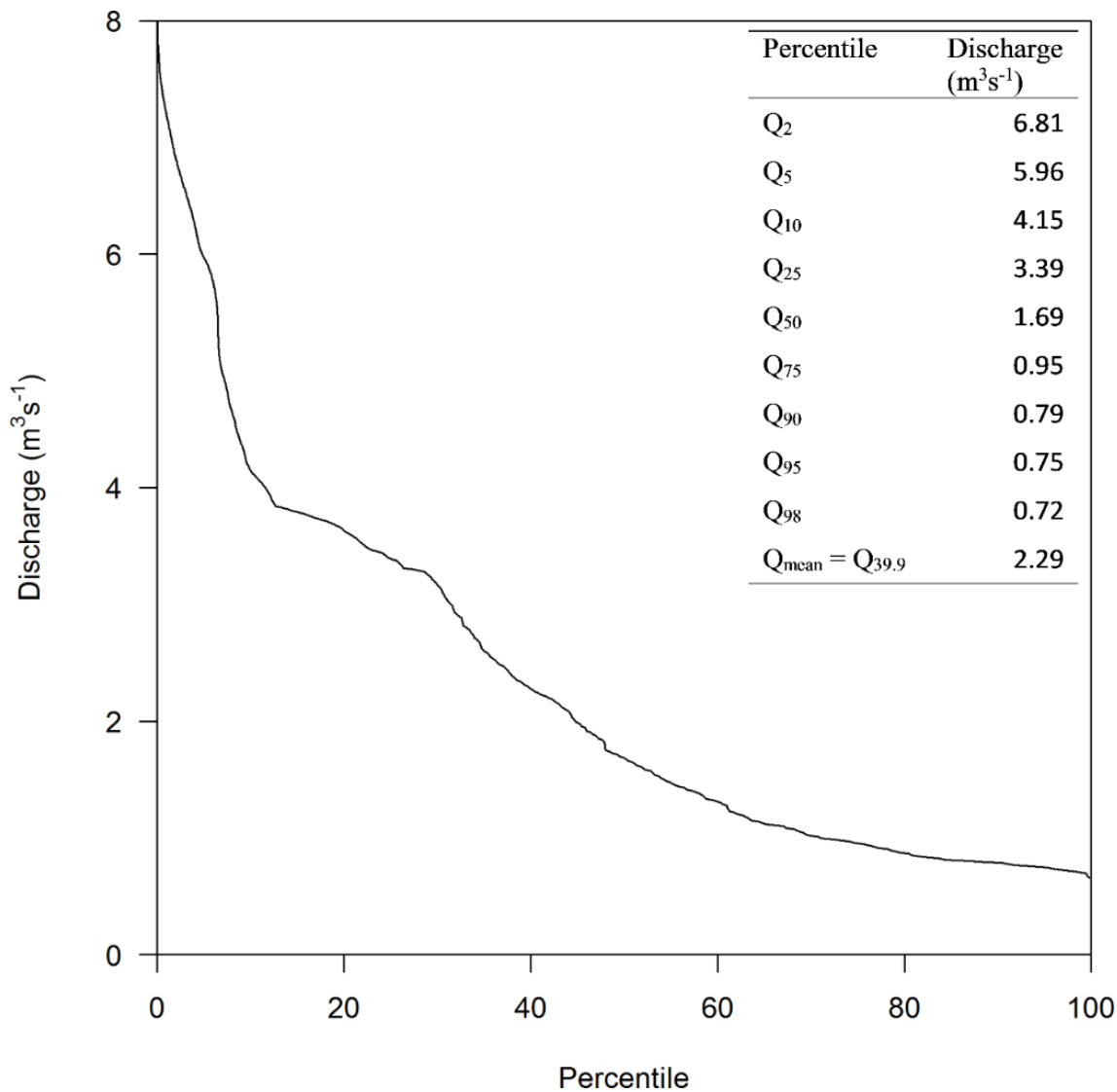


Figure 3-16. Flow duration curve and percentage exceedance flows for the River Lambourn at the Observatory

range of river types (Champion and Tanner, 2000; Clarke, 2002; Clilverd et al., 2013; Naden et al., 2006; Old et al., 2014). Spring and summer growth results in modifications to flow velocity and overall hydraulic resistance, along with reductions in channel capacity, which cause increases in stage. The removal of large volumes of vegetation through cutting increases channel capacity and flow velocity, reducing stage.

Due to the influence of macrophyte growth there is no well-defined curvilinear relationship between stage and discharge (Figure 3-20). Several values of stage exist for distinct discharges. This is most apparent below the 10% exceedance flow of $4.15 \text{ m}^3\text{s}^{-1}$, above which the flow duration curve steepens noticeably (Figure 3-14) and the point distributions of stage against discharge converge to closer relationships. Exceedance flows above 10% are associated with winter and spring months January

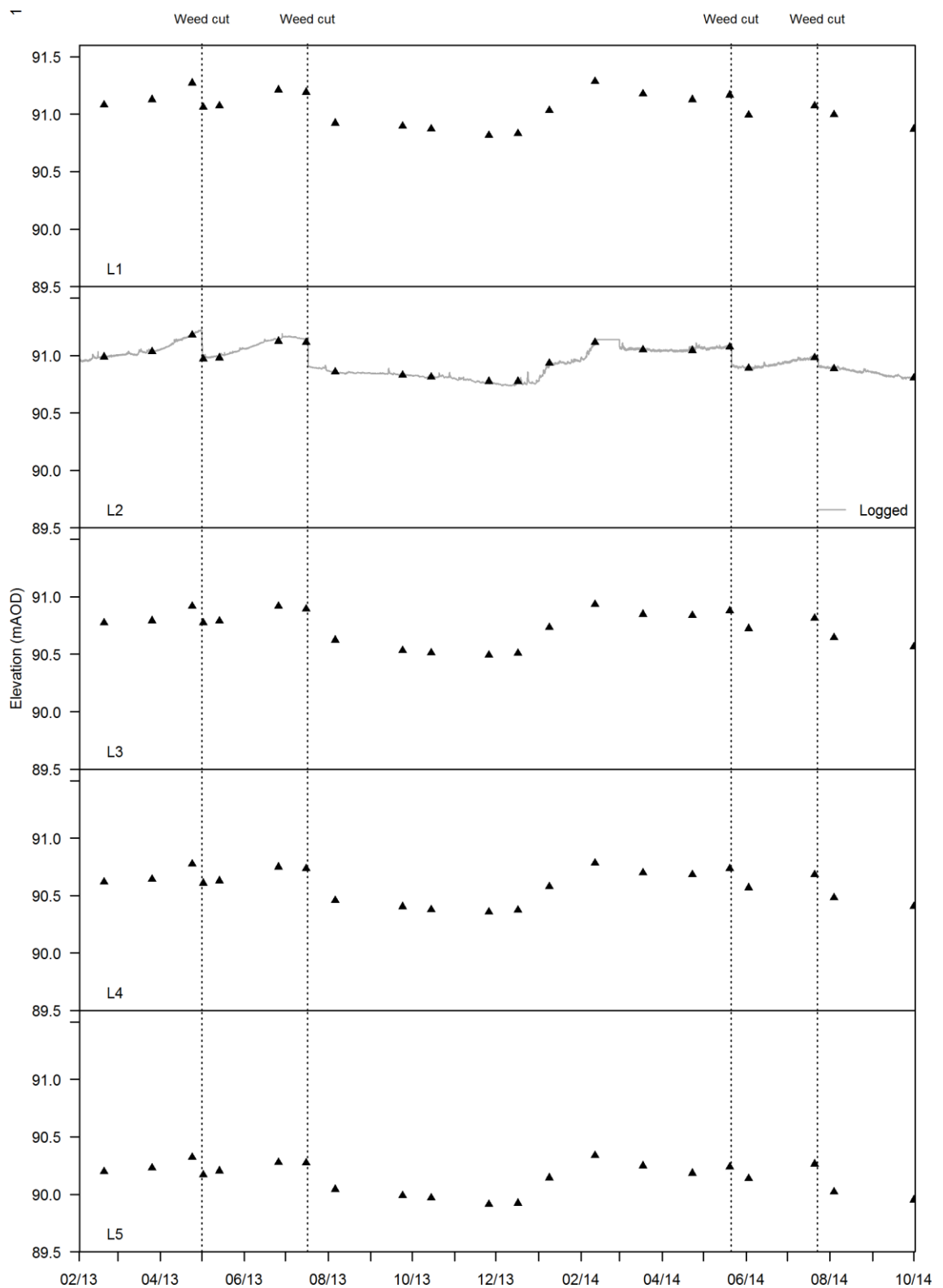


Figure 3-17. Observations of water elevations from stage boards in the River Lambourn L1 – L5 and 15 min logged observations at L2.

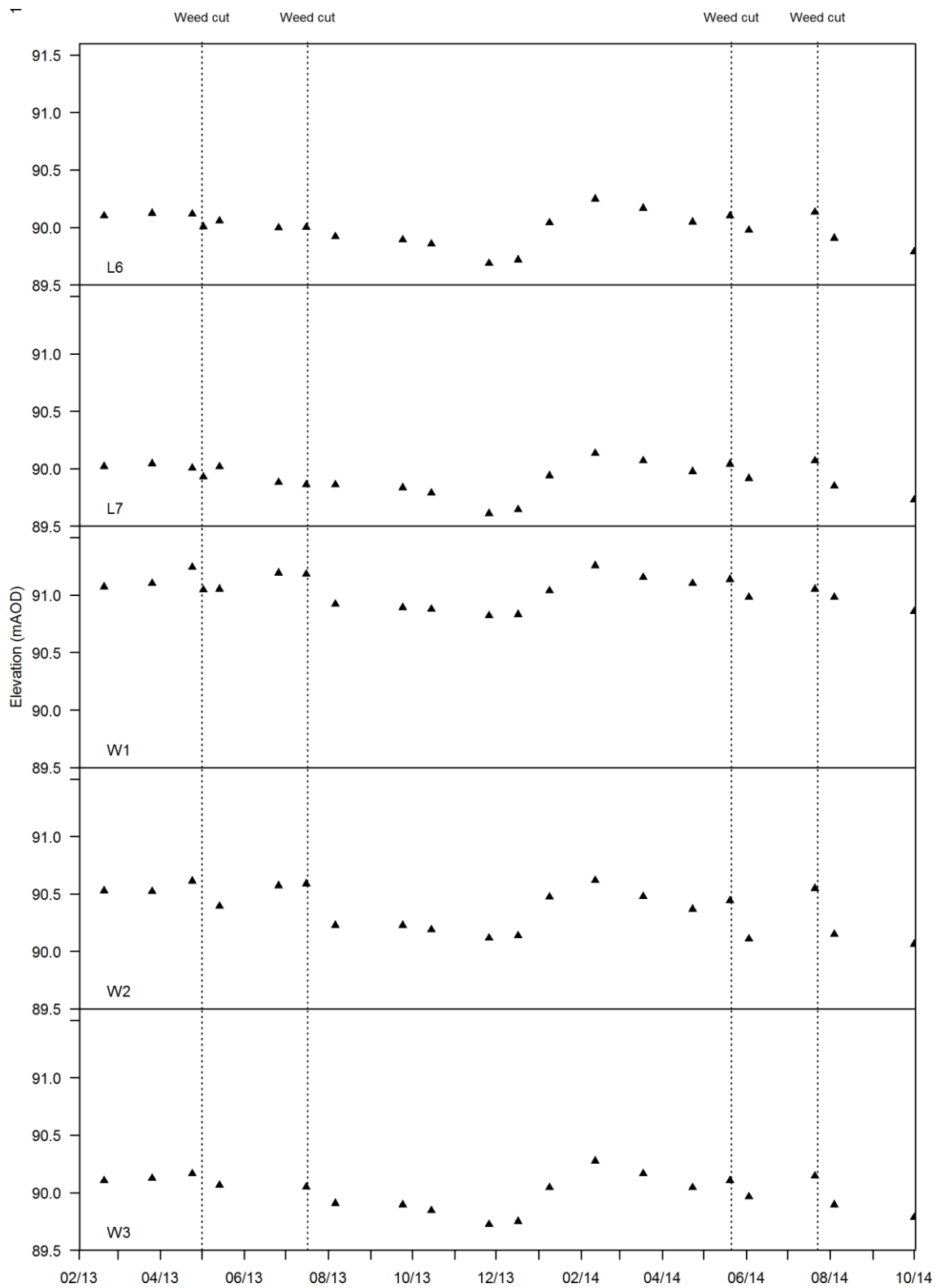


Figure 3-18. Observations of water elevations from stage boards in the River Lambourn L6 – L7 and Westbrook W1 – W3



Figure 3-19. In-channel macrophyte growth and traditional management by cutting with scythes in the River Lambourn

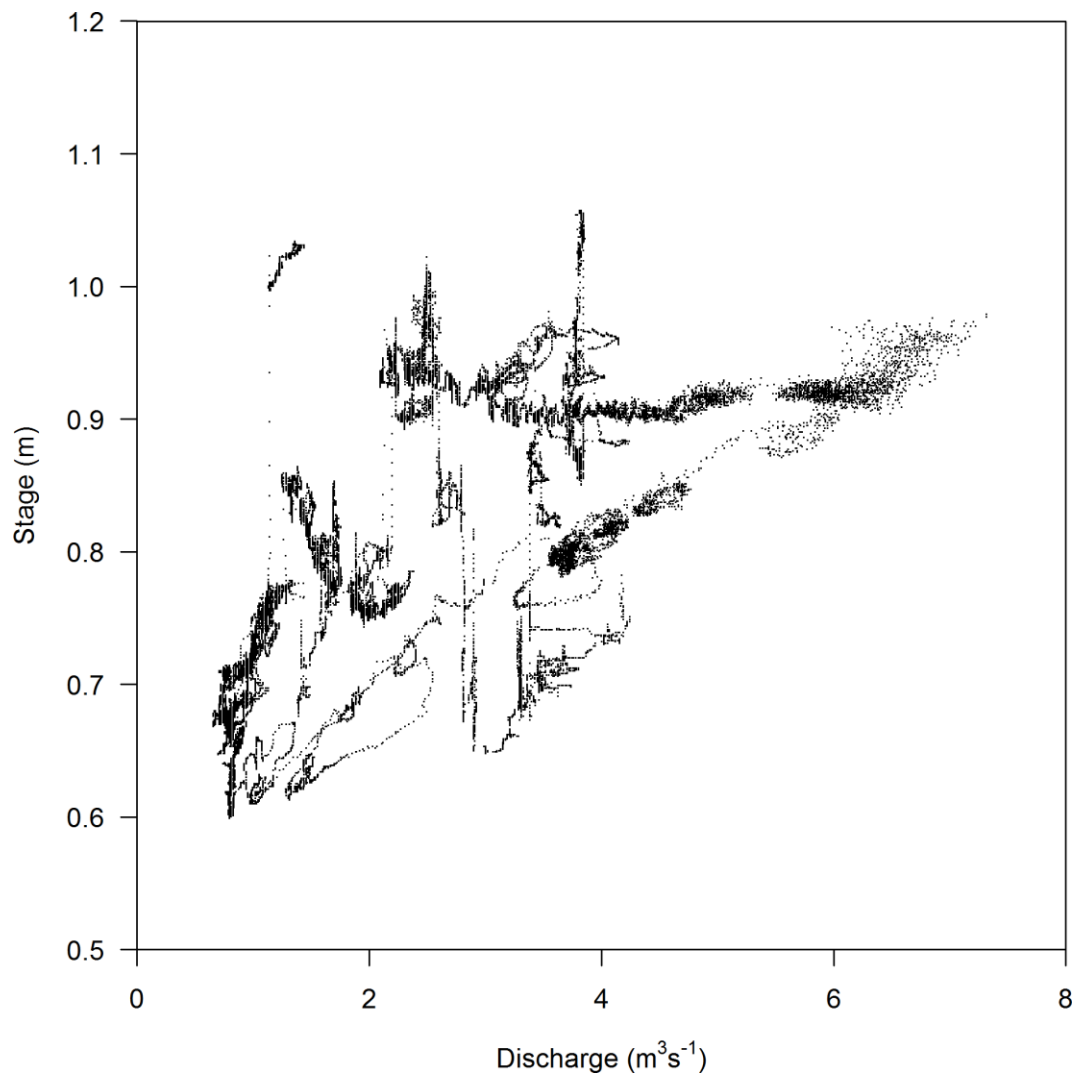


Figure 3-20. Relationship between stage and discharge for the River Lambourn over the monitoring period 1/2/2013 – 1/10/2014

2014 – April 2014, mostly outside of the growing season when the influence of macrophytes will be reduced. Furthermore, the effects of macrophytes may be diminished at high flows due to flattening of the macrophyte stems or removal of vegetation (Chambers et al., 1991; Champion and Tanner, 2000; Clilverd et al., 2013).

3.6 Subsurface peat temperatures

3.6.1 3D peat temperature model

Temperature of the peat was characterised in daylight hours over five days in February 2013 when the difference in surface temperature and groundwater temperature was expected to be pronounced. The north meadow was surveyed between 11-13th February and the south meadow between 18-19th February. An Oakton™ Type T thermocouple probe connected to a thermocouple thermometer was inserted into the peat at 1056 locations on an approximate 5 m × 5 m grid (Figure 3-21). These were georeferenced with Trimble R8™ dGPS. Measurements were taken at depths of 0.15, 0.30, 0.45, 0.60, 0.75, and 0.90 m below ground level resulting in a total of 5109 temperature measurements, with the peat less than 0.90 m thick in places. Dense scrub and watercourse boundaries confined the survey extent.

Temperature data were imported into the 3D visualisation and analysis application Paradigm SKUA 2011.3™. This modelling package allows the integration and analysis of borehole, geophysical and other subsurface data. A 3D grid was constructed which encapsulated the temperature measurement points and had cell dimensions of 1 × 1 m in the horizontal plane and 0.05 m in the vertical direction. The north and south meadows were kept separate. The grid was positioned so that the temperature measurement points coincided with the centre of grid cells. It was deformed to align each layer of cells parallel to the undulating surface topography. Measured temperature values were assigned to the grid cell in which they occurred and, following variogram modelling, were interpolated throughout the gridded volume using ordinary kriging (Webster and Oliver, 2007).

The spherical 3D variogram model was estimated within Paradigm SKUA 2011.3™ by the method of moments. The software includes the capability to include anisotropy in the variogram model. This is where the expected squared difference between a pair of observations is a function of both the length and the direction of the vector which separates the measurement locations (Banerjee et al., 2014). In contrast, isotropic models assume that the variogram is purely a function of the length of the vector.



Figure 3-21. Temperature survey measurement locations

In the horizontal plane, any directional-dependence appeared to be caused by a single feature in the northern meadow. The temperature was less variable parallel to this than perpendicular to it. Since the feature dissected the meadow, it was not possible to represent this anisotropy by a geometric distortion (stretching or contraction) of the variogram range in a single direction. Any attempt to do so would have led to artefacts in the predicted temperature surface. There was a clear difference between the variograms in the horizontal and vertical planes. Therefore the estimated model was isotropic in the horizontal plane with geometric anisotropy in the vertical plane (Webster and Oliver, 2007). The estimated nugget and sill variances are $0.01\text{ }(^{\circ}\text{C})^2$ and $0.52\text{ }(^{\circ}\text{C})^2$, respectively. The range of the model is 35.5 m in the horizontal direction and 0.3 m in the vertical direction.

3.6.2 Temperature distribution

Overall temperature ranged between 2.1 and 10.3°C in the peat, with a mean temperature of 6.4°C ($\sigma = 1.7^{\circ}\text{C}$). In the north meadow the mean temperature was 6.5°C ($\sigma = 1.76^{\circ}\text{C}$). In the south, mean temperature significantly was smaller at 6.0°C ($\sigma = 1.32^{\circ}\text{C}$), using the two-sample t-test for unequal variances, $t(4172) = 1.96$,

$p < 0.001$. The model shows discrete warm temperature anomalies ($>9^{\circ}\text{C}$) which correlate with historical channels (Figure 3-22), which have infilled and are no longer present. In the north meadow the anomalies largely correspond to the path of a sinuous relic channel (D – D', Figure 3-22ii), whilst the main anomaly in the south meadow lies

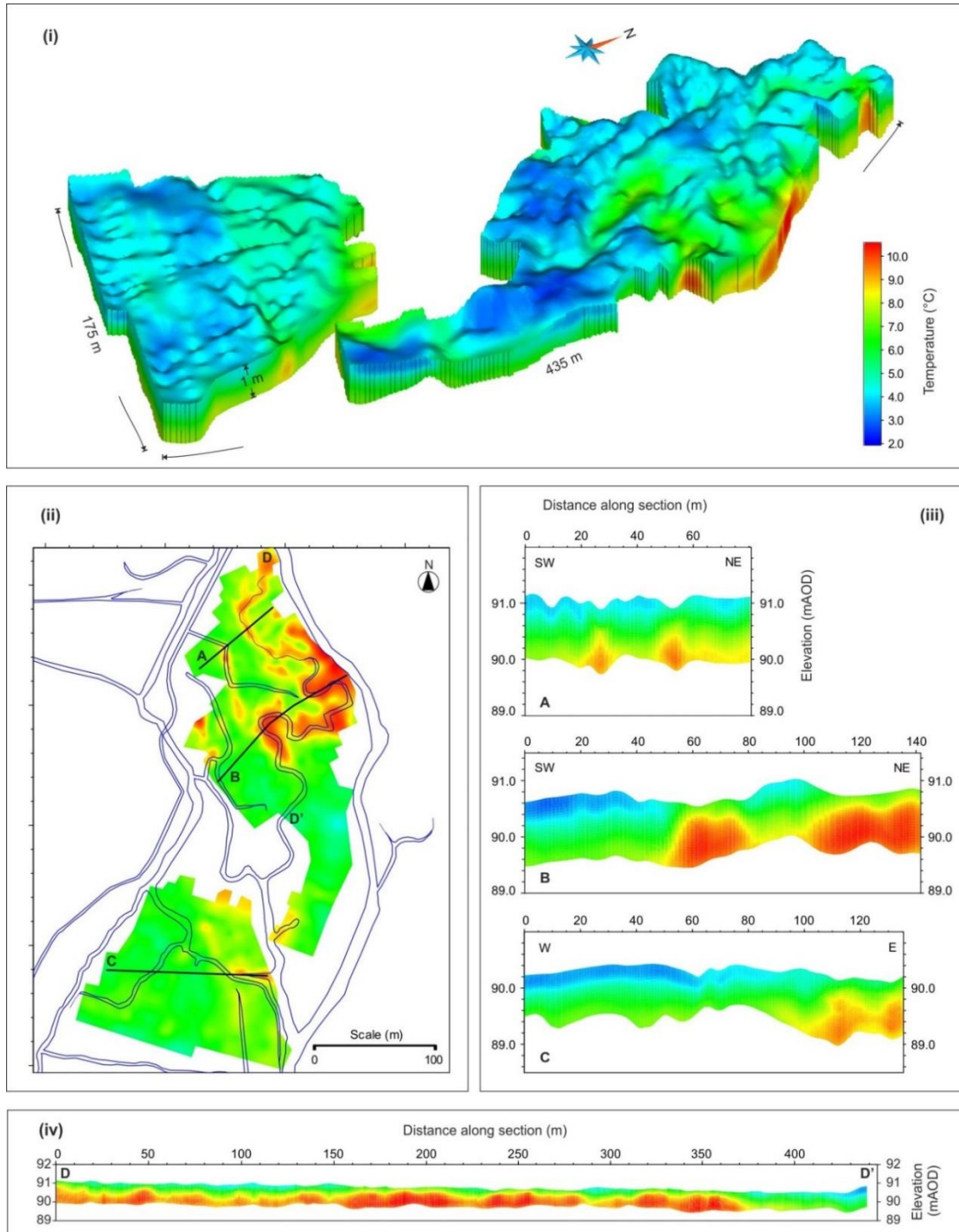


Figure 3-22. Subsurface temperature across wetland: (i) 3D temperature model with X20 vertical exaggeration, (ii) plan view of temperature at 0.7 m bgl with historic channels, (iii) temperature cross-sections across wetland, and (iv) cross-section along infilled relic channel.

within a relic channel draining to the Westbrook. The anomalies are tightly constrained with heat only propagating 5-10 m laterally from the channel centre.

Temperature gradients vary widely across the site, and cross-sections through the model show they are relatively constant with depth at some of the strongest temperature anomalies (Figure 3-22iii). Overall the mean temperature increases from 4.7 °C at 0.15 m to 7.9 °C at 0.9 m, while σ drops from 1.3 to 0.9 °C. The decreasing σ with depth reflects a decreasing range from between 2.1 °C and 10.0 at 0.15 m, down to between 5.7 and 10.3 °C at 0.9 m.

A warm temperature anomaly is located at the source of the main relic channel in the north meadow, with further anomalies spread along its course (Figure 3-22iv). Anomalies within the peat frequently extend toward the surface, but then gradually dissipate with distance. Greatest changes in temperature occur at shallow depths. Below 0.7 m warm anomalies are persistent along the entire channel length before dispersing at around 370 m from the source.

Air temperature ranges during the peat temperature surveys were -0.3 to 2.6 °C and -5.0 to 10.6 °C for the north and south meadows respectively (Figure 3-23). These fluctuations were significantly dampened through the peat depth, with total variations of only 0.8, 0.3, and 0.1 °C at 0.1, 0.3 and 0.5 m bgl, respectively.

Groundwater temperature within the gravels ranged from 9.4 to 10.2 °C between piezometers in the north meadow, with the exception of piezometer 6 which was around 7.3 °C. There appears to be a temperature gradient across the meadow with the warmest water located further north. Within the south meadow the gravel groundwater was cooler at 8.3 to 9.0 °C.

The River Lambourn temperature varied between 7.7 and 8.2 °C, with a mean of 7.9 °C, during the north meadow survey. During the south meadow survey the variation was 7.3 to 9.4 °C, with a mean of 8.4 °C.

3.7 Wetland water levels

3.7.1 Piezometric elevations

Gravel and peat head elevations in the wetland are reminiscent of the pattern of channel stages throughout the monitoring period (Figures 3-24 to 3-27). Periods of high head elevation through February 2013 – July 2013 and January 2014 – July 2014 are contrasted by low head elevations in the remaining months. Sharp drops in head coincide with the decreases in stage associated with the weed cuts. The similarity in responses

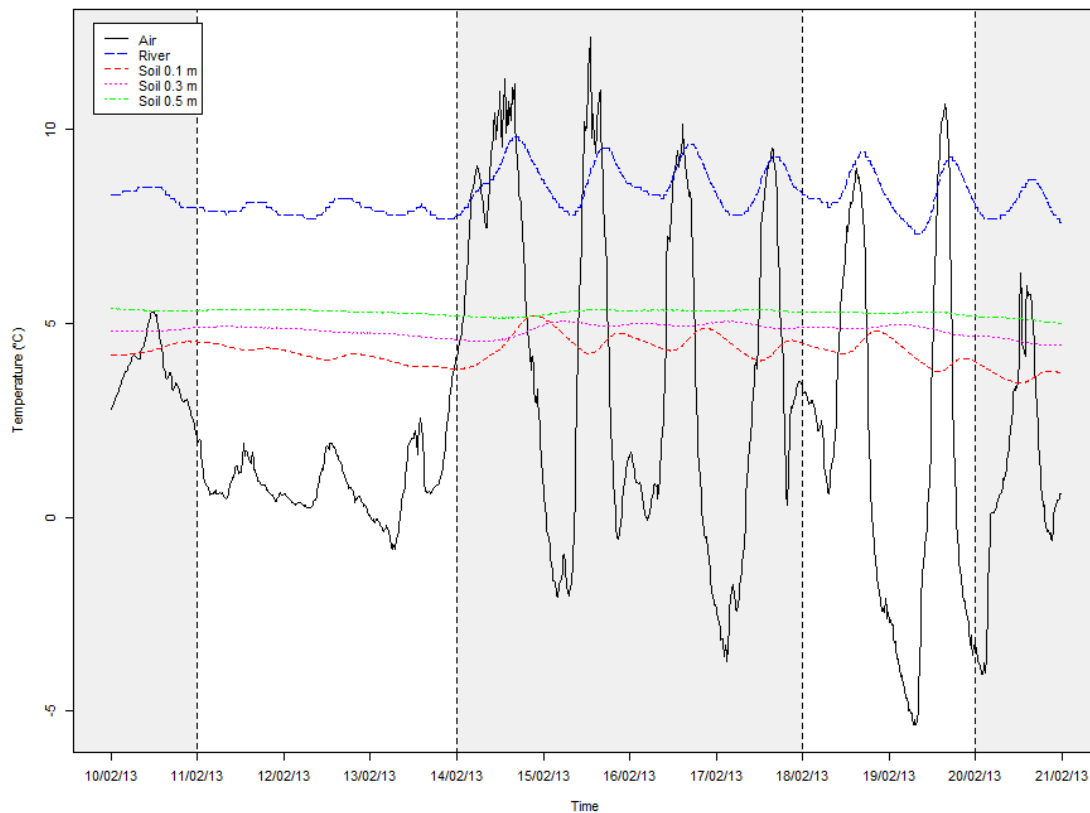


Figure 3-23. Records for River Lambourn temperature at WQ, plus air and soil temperature at AWS. A grey background indicates records not relating to the period of the temperature spatial survey.

to rain events and weed cuts indicates the peats and gravels are in hydraulic connectivity, and that they are connected to the main channels.

Generally peat heads are below gravel heads, with reversals at periods of exceptional high water levels around March 2013 – April 2013 and January 2014 – February 2014. At these times the peat head elevations plateau while the gravel peaks. This is most noticeable in the piezometers installed to target temperature anomalies (14-19), especially at site 14, yet also in other piezometer sites in the north meadow (1-5). In south meadow piezometers (9-13) gravel heads rarely rise above peat heads, only doing so in February 2014. The piezometer pair at site 6 shows an exception, with peat heads plateauing over gravel heads throughout periods of high flow. An exaggerated response to rain events at periods of low water levels is apparent in some peat

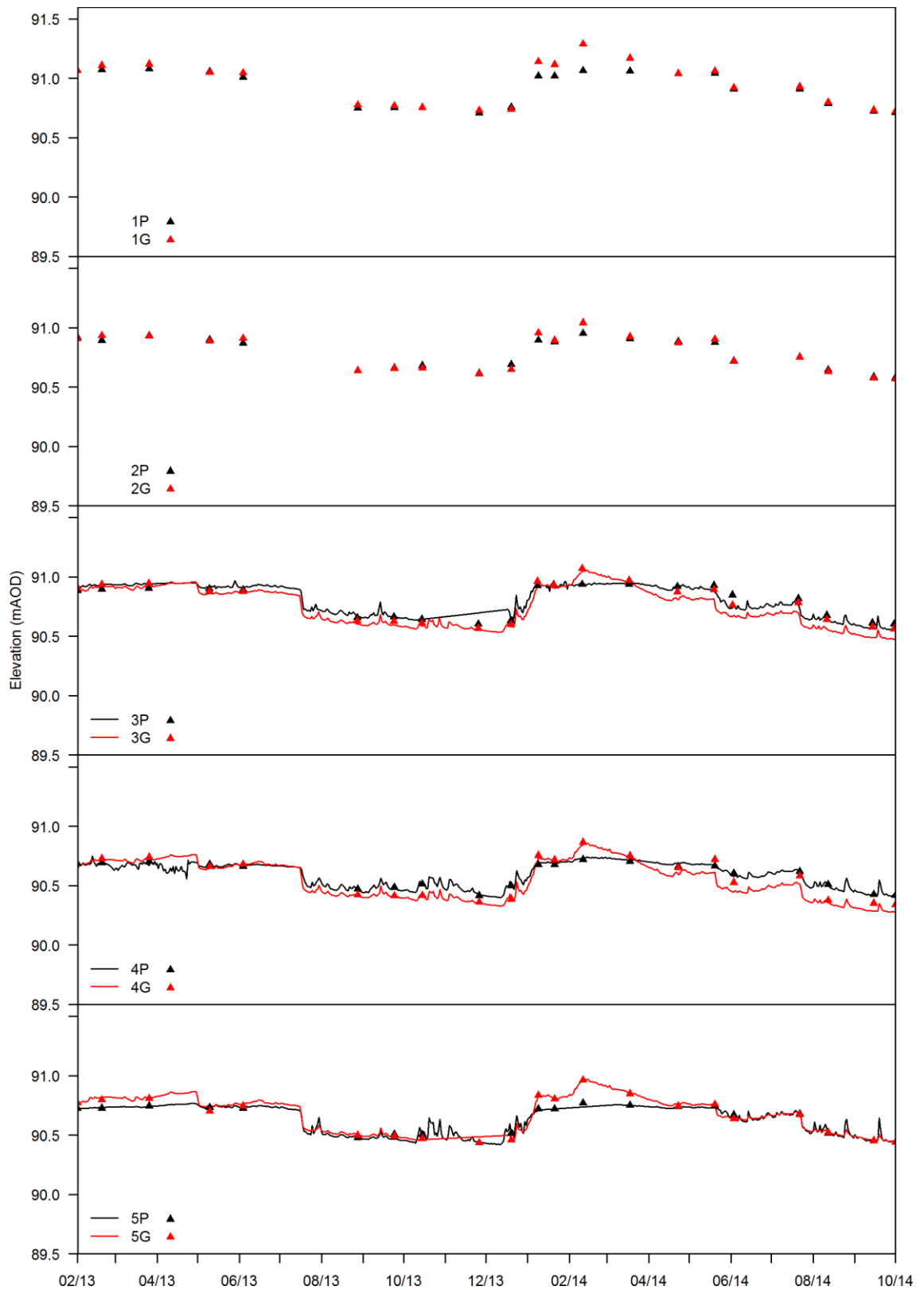


Figure 3-24. Dipped and 15 min logged head elevations for peat (P) and gravel (G) piezometer sites 1-5

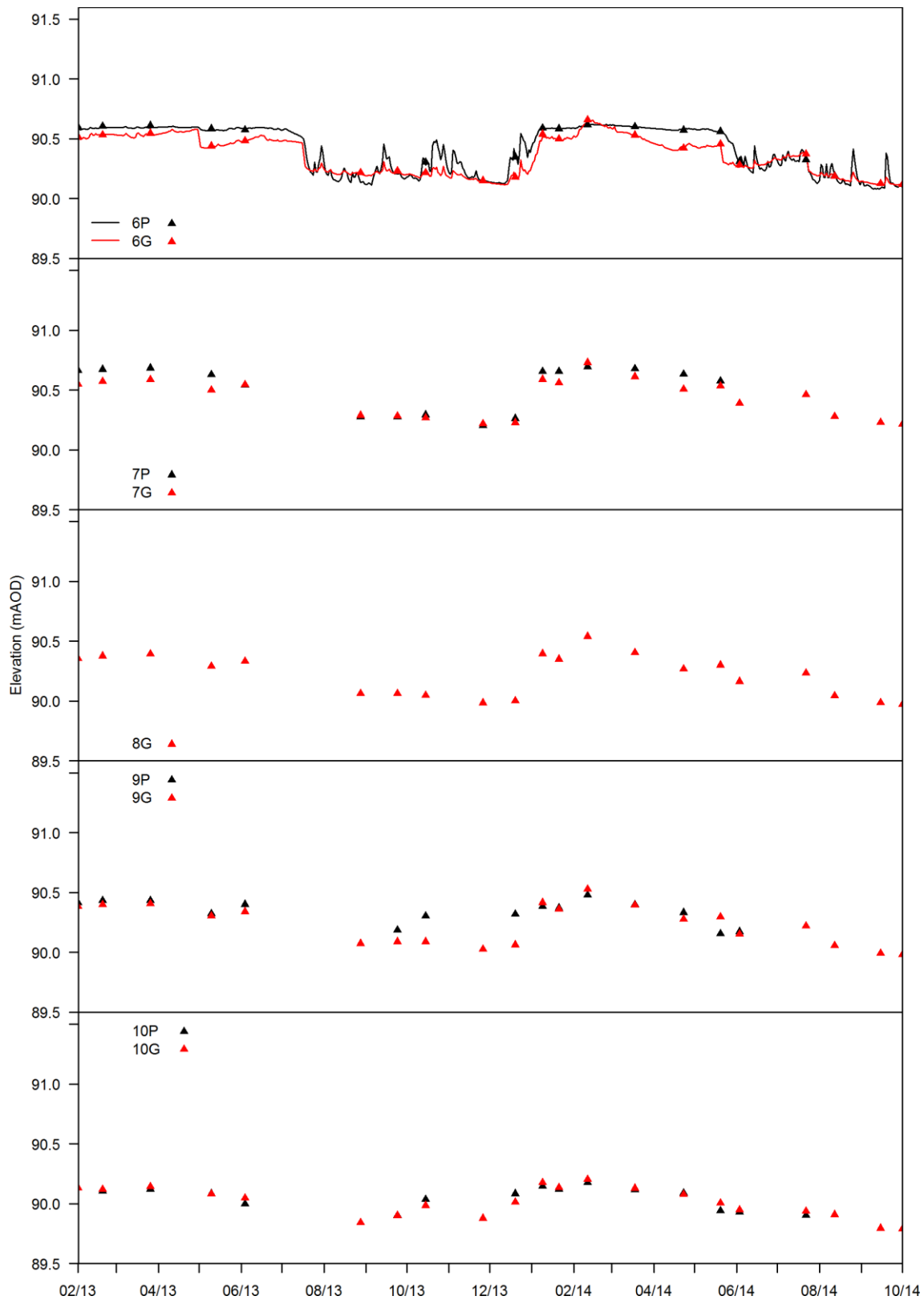


Figure 3-25. Dipped and 15 min logged head elevations for peat (P) and gravel (G) piezometer sites 6-10

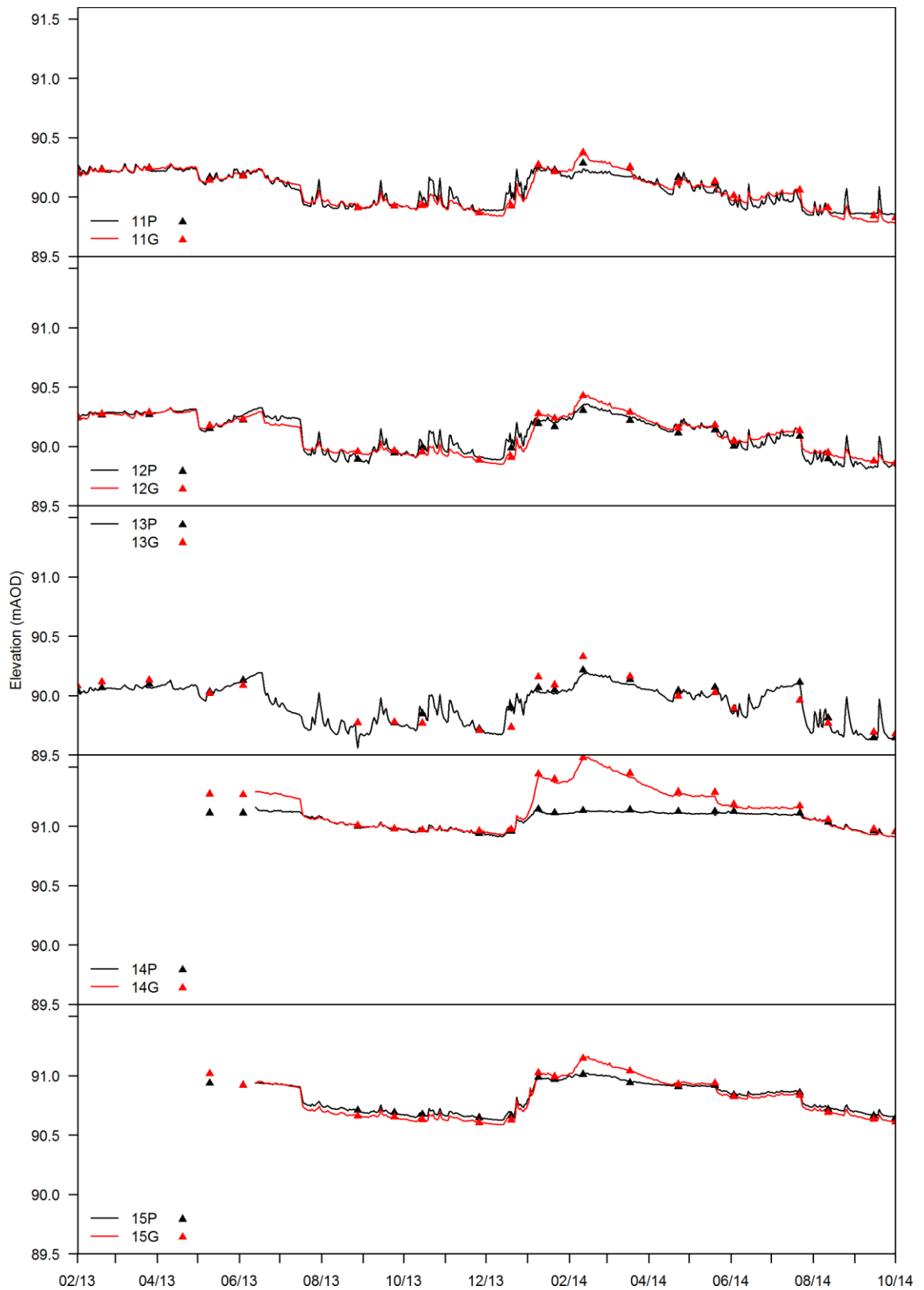


Figure 3-26. Dipped and 15 min logged head elevations for peat (P) and gravel (G) piezometer sites 11-15

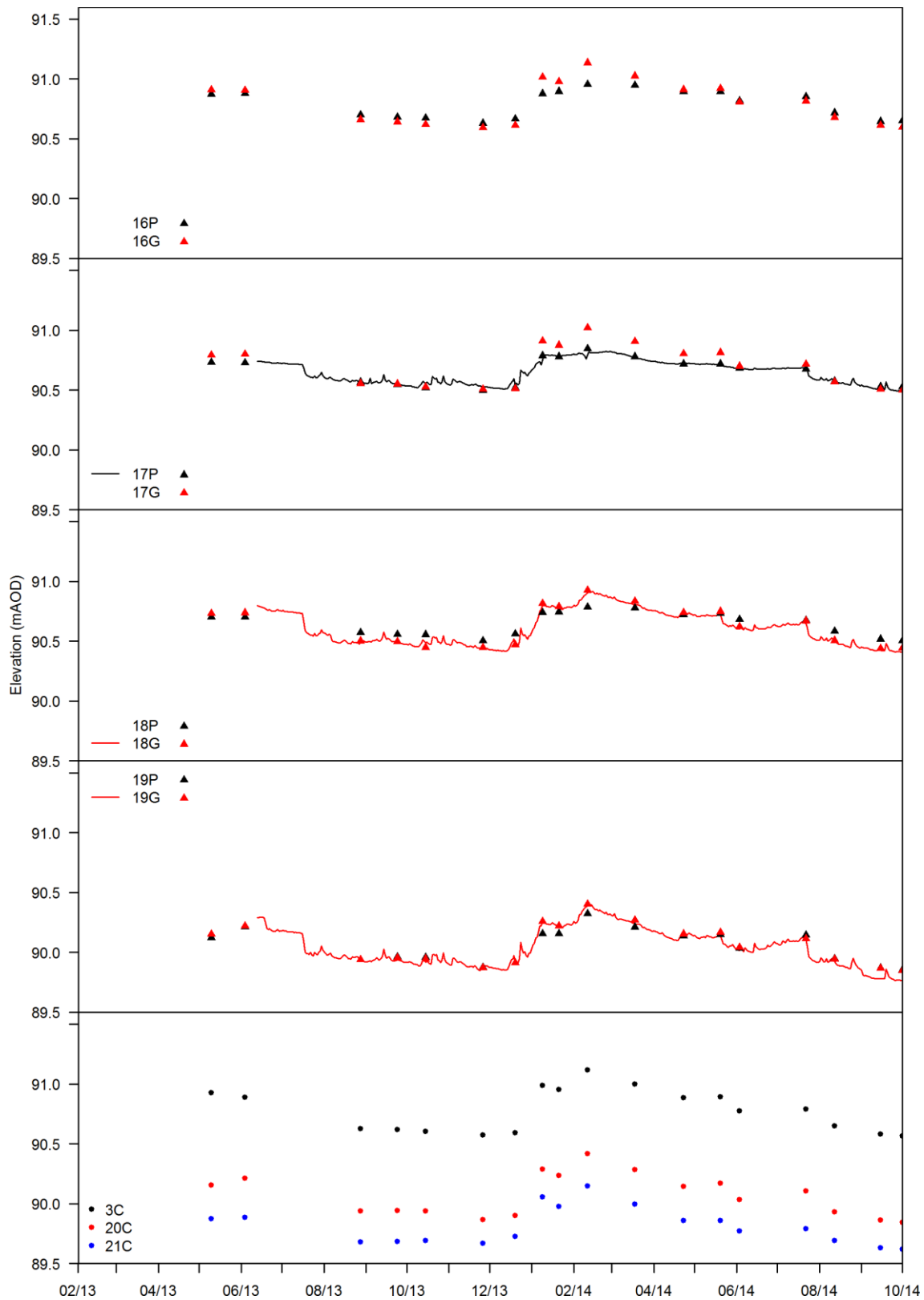


Figure 3-27. Dipped and 15 min logged head elevations for peat (P) and gravel (G) piezometer sites 16-19 and chalk piezometer sites 3, 20 and 21

piezometers (6P, 11P, 12P, 13P). These piezometers are from a pre-existing array installed with the slotted screen extending above ground level. This is in contrast with locations 14P-19P where bentonite was used to seal new piezometers with closed screens above ground level. The sharp observed responses may be a reflection of direct influx of water from the surface during rain events. The plateau of high head elevation at 6P may thus be due to surface water above the open level of the piezometer.

Chalk head elevations display a comparable cycle of high and low level periods. Differences between chalk piezometers are consistent through the monitoring period. There is greater similarity between the pattern of chalk and gravel heads rather than peat. This is evident in the sharp peak in February 2014.

3.7.2 Piezometric gradients

Vertical head gradients differ spatially and over time through the monitoring period (Figures 3-28 to 3-32). At sites 14-19, where piezometers are drilled into the relic channels containing the warm temperature anomalies, vertical gradients differ to the rest of the site. Before their installation in May 2013, and through a period of high water levels, upward gradients existed in the northeast and centre of the north meadow, and to the east and west edges of the south meadow. Elsewhere predominantly downward gradients were present. In May 2013 upward gradients occurred at sites 14-19, and also at sites 5 and 12, persisting into June 2013 with the addition of sites 1, 2, 4, 7 and 10. Through the period of low water levels (August 2013 – December 2013) there was a shift to downward gradients. This began with sites 15-18 and 3-4 in August and spread to the remaining piezometer sites, with a few exceptions at 1, 5, where vertical gradients fluctuated, and at 14 where an upward gradient largely persisted. River Lambourn and Westbrook levels were generally higher than peat and gravel heads at the beginning of this period, dropping to equivalent levels by December 2013. A swift reversal upwards in gradients was conspicuous through January 2014 – February 2014. Vertical gradients at sites 14-19 were visibly higher than at other sites through this period of high water levels. Gradually these dropped through the remainder of the monitoring period, with predominantly downward gradients occurring particularly around sites 2-5 and 15-18. Large downward gradients were evident at 6 and 7, though, at sites 1 and 14 upward gradients continue.

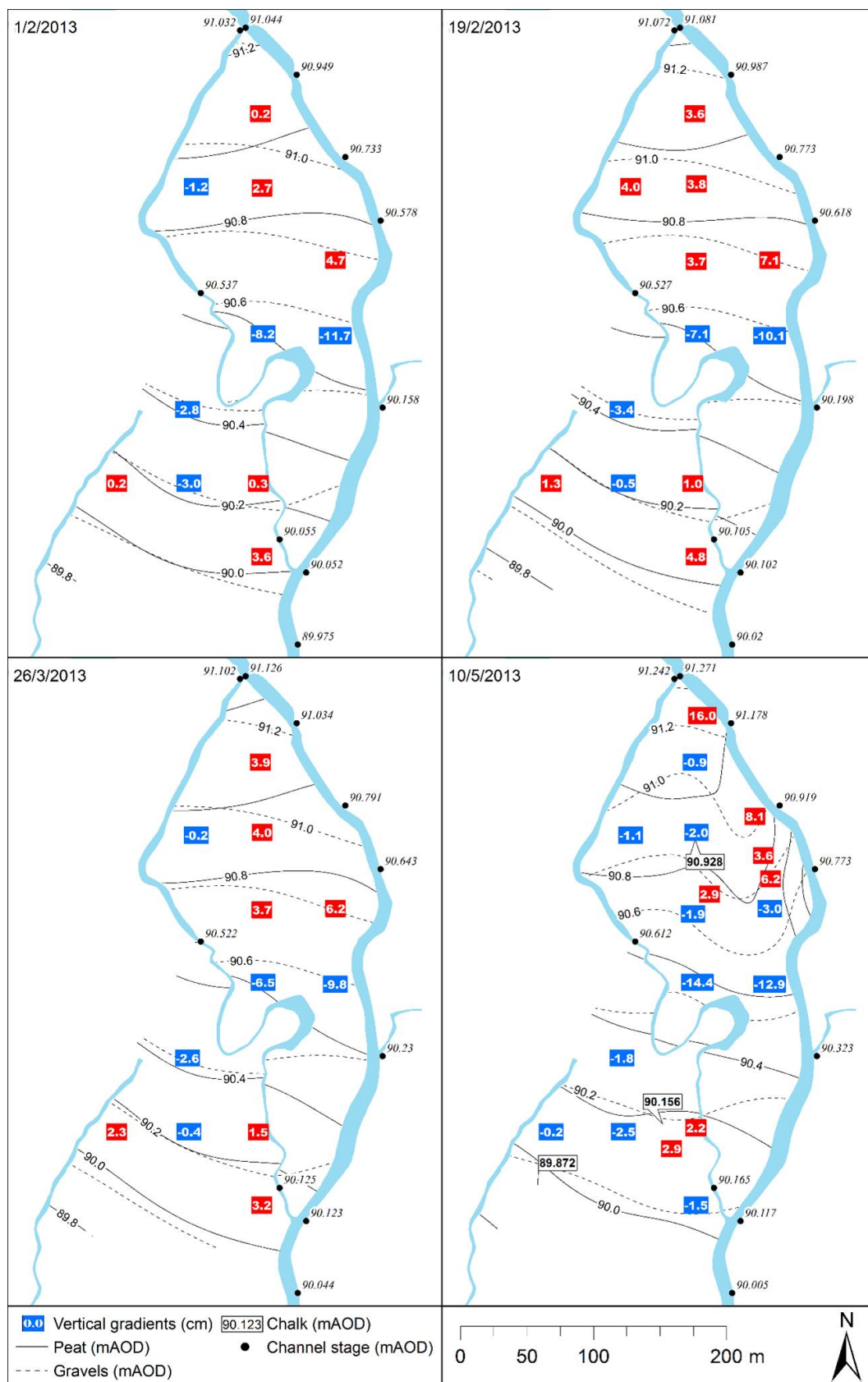


Figure 3-28. Groundwater head map for peat and gravels with vertical gradients (red upward blue downward), chalk heads, and channel stage elevations February 2013 – May 2013

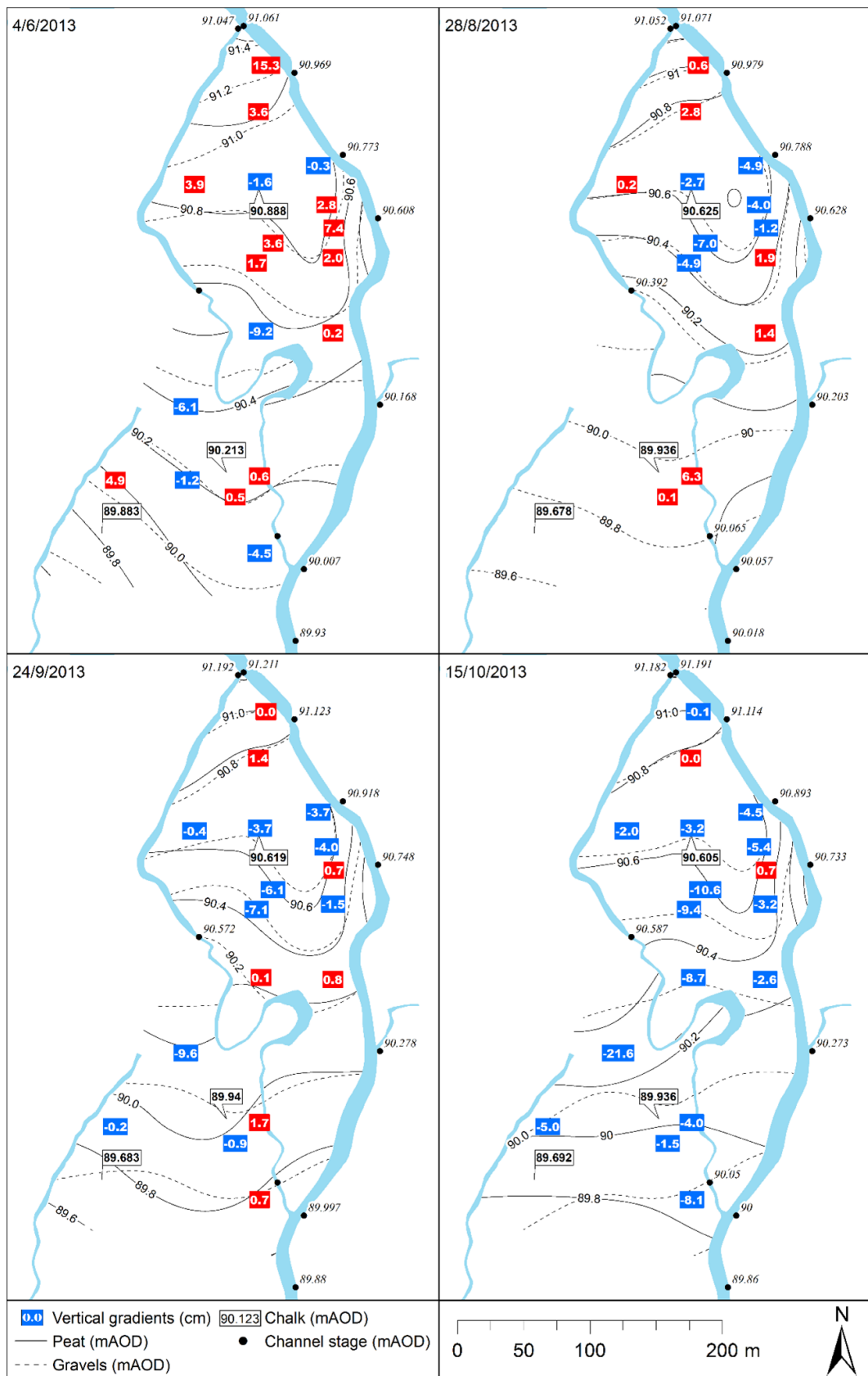


Figure 3-29. Groundwater head map for peat and gravels with vertical gradients (red upward blue downward), chalk heads, and channel stage elevations June 2013 – October 2013

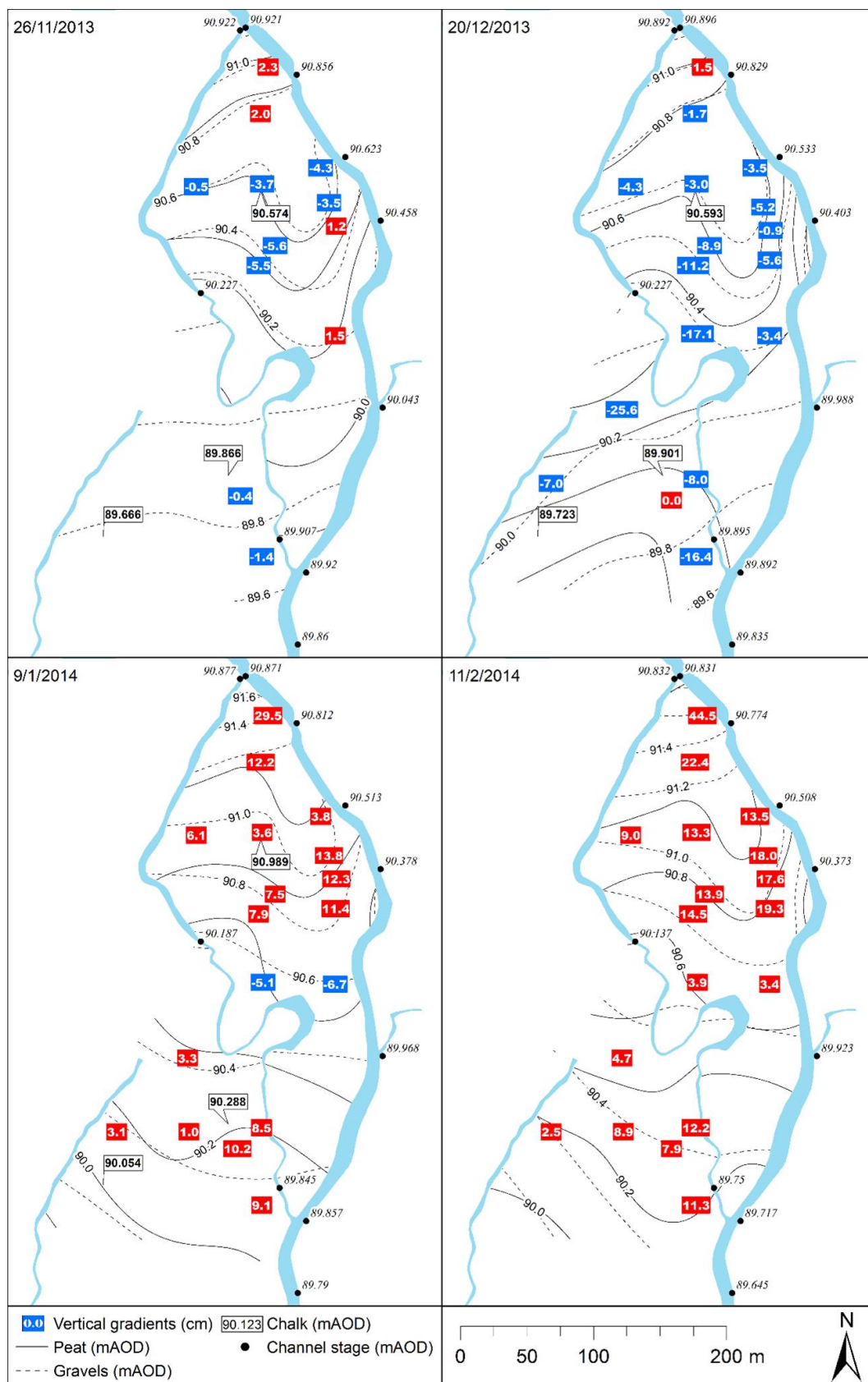


Figure 3-30. Groundwater head map for peat and gravels with vertical gradients (red upward blue downward), chalk heads, and channel stage elevations November 2013 – February 2014

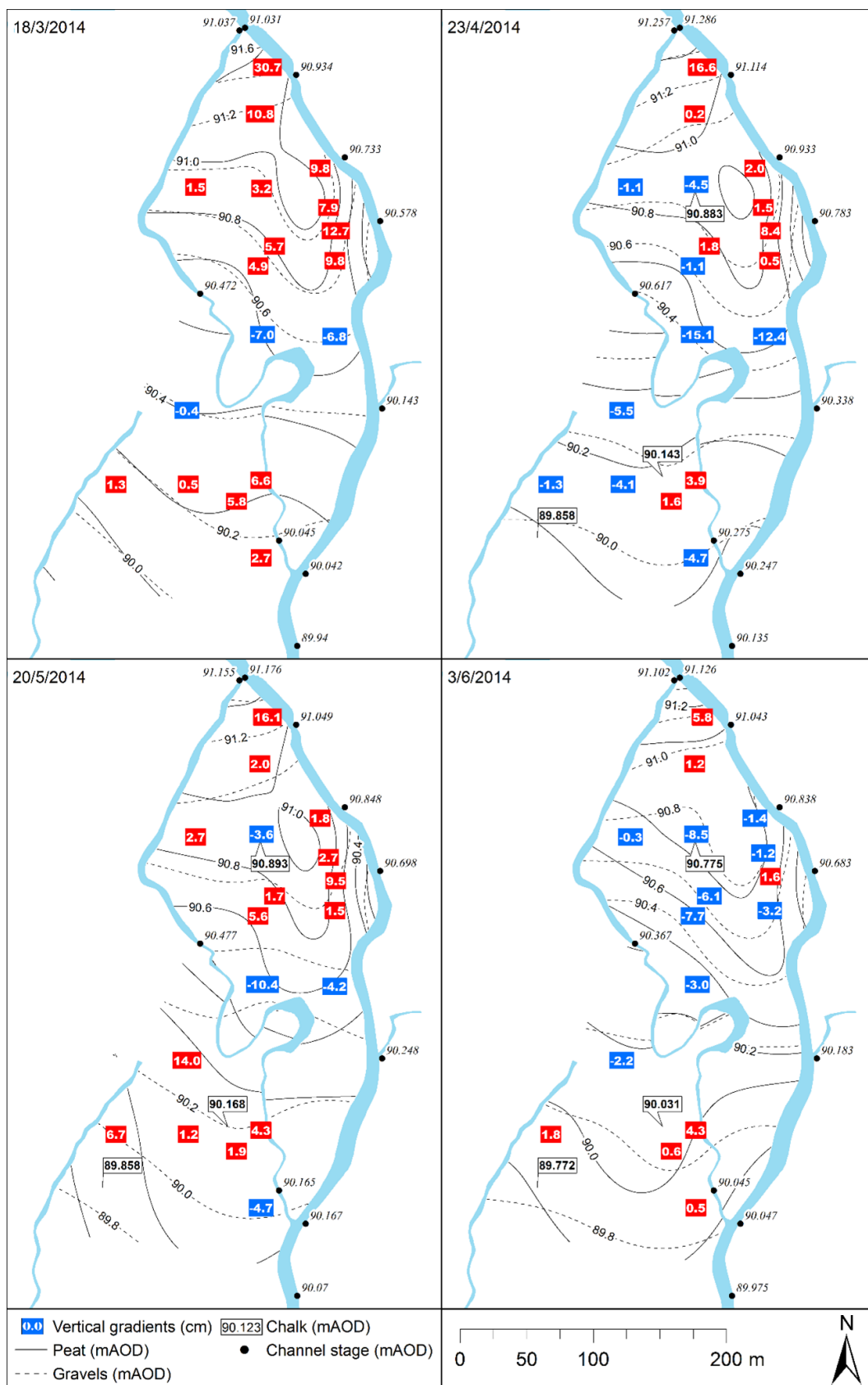


Figure 3-31. Groundwater head map for peat and gravels with vertical gradients (red upward blue downward), chalk heads, and channel stage elevations March 2014 – June 2014

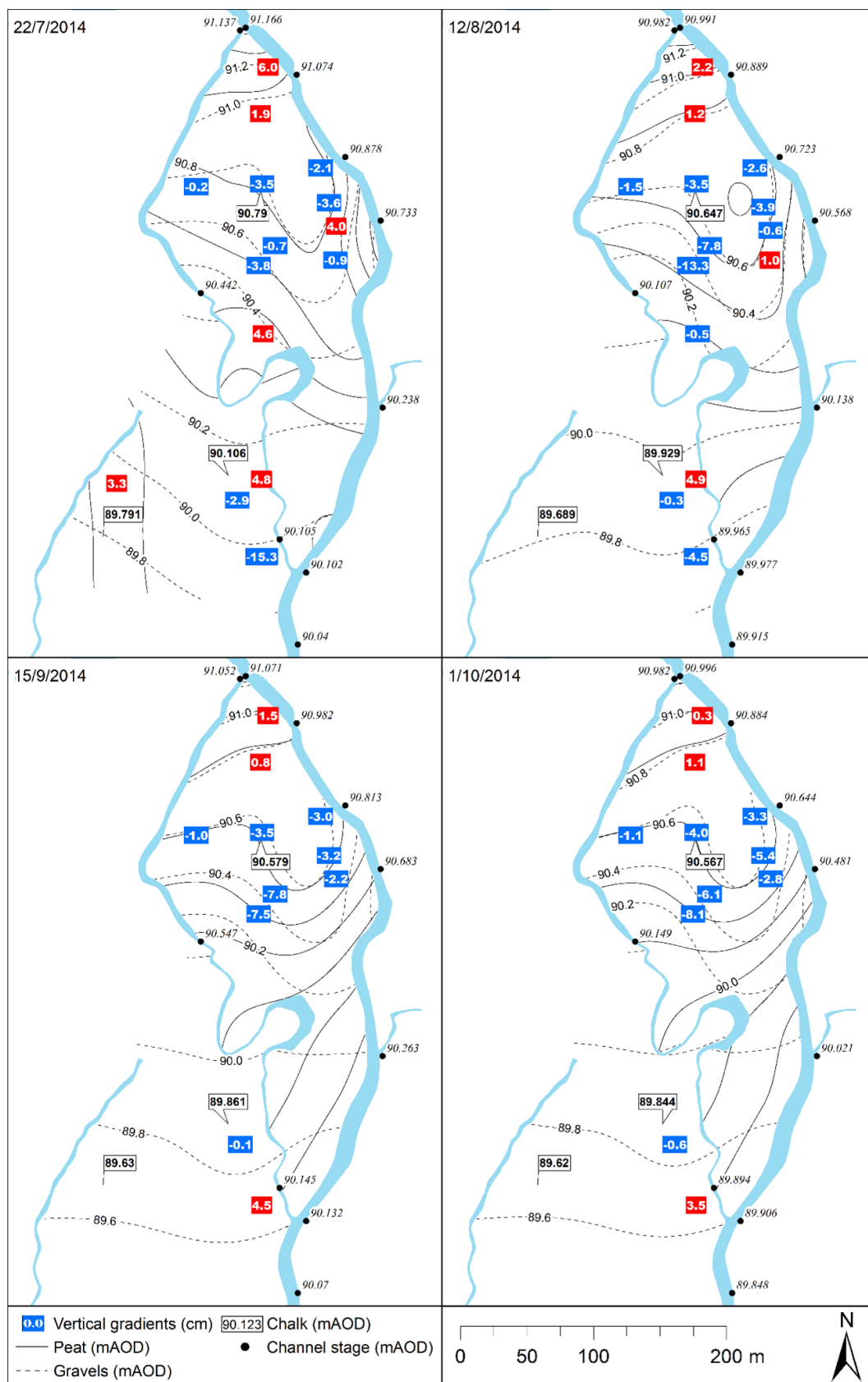


Figure 3-32. Groundwater head map for peat and gravels with vertical gradients (red upward blue downward), chalk heads, and channel stage elevations July 2014 – October 2014

Generally, groundwater head contours follow a south-westerly direction at an overall gradient of 0.003 in peat, gravel, and chalk. There is a noticeable area of groundwater mounding around sites 15-18 that persisted through the monitoring period, and coincides with the main temperature anomaly. Chalk heads corresponded closely with gravel head contours throughout the monitoring period. Vertical gradients at sites 14-19 generally appear to correspond with changes in chalk head elevation, with relatively high chalk heads coinciding with upward gradients and low chalk heads with downward gradients. Conversely, groundwater levels at sites 1-13 from the pre-existing array correspond to water levels in the Westbrook and River Lambourn. When channel water levels were high, upward vertical gradients often occurred, and when low downward gradients existed. However, exceptions to this generalisation exist, especially at the extremes of high or low water level periods.

3.7.3 Piezometer movement with peat expansion and contraction

Piezometers were not anchored to bedrock, with the possible consequence that datums for water levels, taken at the top of piezometers, could move with the expansion and contraction of the peat due to saturation. Though Price and Schlotzhauer (1999) suggested that shallow (<0.5 m) peat generally possesses low compressibility, piezometer elevations were surveyed at periods of low (November 2013) and high (April 2014) water table level (Section 3.7.1) to ascertain any vertical movement. Surveys were carried out using Trimble 5600 DR™ total station and Trimble R8™ dGPS. Three total station setups were positioned with dGPS based on line of sight to fixed benchmark points at the Westbrook sluice and River Lambourn stone bridge, and to enable best coverage (Figure 3-33). Selected piezometers were included in the surveys based on line of sight to total station setups (Table 3-2).

Results from the surveys show differences can be grouped dependent on instrument setup (Table 3-3). Mean differences for setups 1, 2, and 3 are 0.019, 0.000 and 0.006 m, respectively, which indicate systematic error and are likely due to inaccuracies in the Trimble R8™ GPS receiver due to obstructions, satellite geometry or atmospheric conditions (Trimble, 2003). Variance around the means is not more than 0.003 m, within the accuracy for the Trimble 5600 DR™ total station (Trimble, 2005). Hence, piezometer movement due to peat compressibility with saturation was discounted.

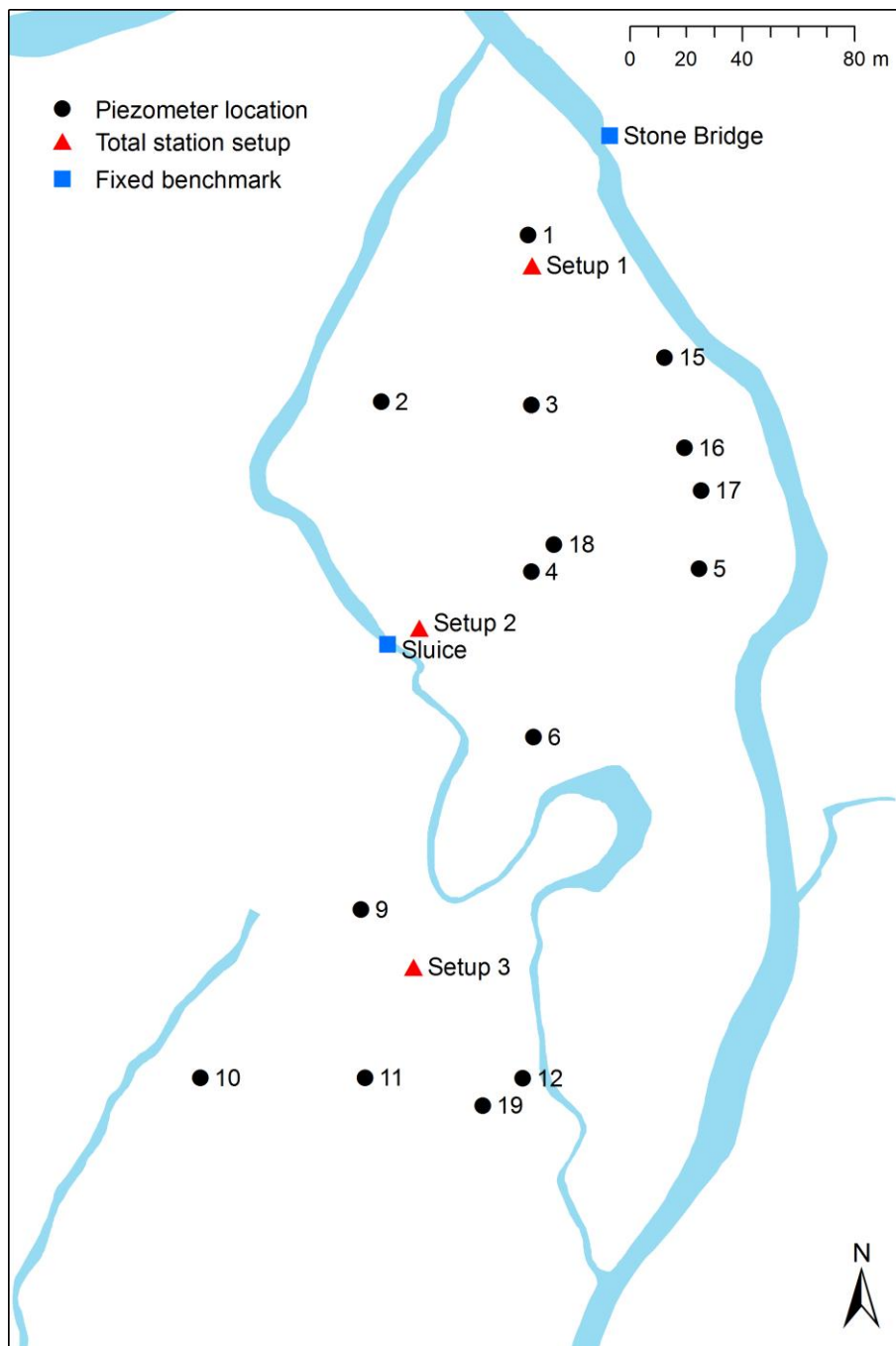


Figure 3-33. Instrument setup locations for piezometer datum surveys

Table 3-2. Instrument setup schedule for piezometer datum surveys

Setup	Foresight	Backsight	Piezometer locations
1	Stone Bridge	Stone Bridge	1, 2, 3, 15, 16, 17
2	Sluice	Stone Bridge	4, 5, 6, 18
3	Sluice	Sluice	9, 10, 11, 12, 19

Table 3-3. Surveyed elevations for peat (P) and gravel (G) piezometers in November 2013 and April 2014 and absolute difference in elevations

Piezometer	Nov-13 elevations (mAOD)	Apr-14 elevations (mAOD)	Absolute difference (m)
1G	92.129	92.147	0.018
1P	91.321	91.339	0.018
2G	91.809	91.828	0.019
2P	91.372	91.390	0.018
3G	91.569	91.589	0.020
3P	91.307	91.326	0.019
4G	91.166	91.166	0.000
4P	90.995	90.997	0.002
5G	91.445	91.443	0.002
5P	91.426	91.425	0.001
6G	91.212	91.213	0.001
6P	91.164	91.165	0.001
9G	91.014	91.007	0.007
9P	91.241	91.235	0.006
10G	90.891	90.885	0.006
10P	91.088	91.080	0.008
11G	90.998	90.992	0.006
11P	91.087	91.082	0.005
12G	91.086	91.078	0.008
12P	90.841	90.835	0.006
15G	91.481	91.500	0.019
15P	91.423	91.441	0.018
16G	91.360	91.380	0.020
16P	91.372	91.391	0.019
17G	91.253	91.276	0.023
17P	91.177	91.200	0.023
18G	91.145	91.145	0.000
18P	91.140	91.137	0.003
19G	90.770	90.764	0.006
19P	90.704	90.699	0.005

3.8 Wetland hydrochemistry

3.8.1 Water sampling and analysis

Groundwater samples were collected for chemical analysis in May 2013, after installation of the new piezometers, from all gravel and chalk piezometers. Samples were taken once stable field measurements were obtained for dissolved oxygen (DO), pH, specific electrical conductance (SEC), redox potential (Eh) and temperature. These measurements were collected using Mettler-Toledo™ probes contained within a flow-through cell to inhibit any contact with the atmosphere. Samples from all peat piezometers were obtained following a single purge, as the transmissivity of the peat was too low to sustain continuous abstraction with a peristaltic pump (0.1-0.5 Lmin⁻¹). Surface water samples were also collected from the River Lambourn and West Brook.

Laboratory sample preparation and analysis were performed following the procedure outlined in Neal et al. (2011). Major anions and dissolved metals were determined via Dionex™ liquid chromatography and ICP-OES, respectively. Alkalinity was determined by titration with hydrochloric acid. Dissolved organic carbon (DOC) analysis was undertaken with a Thermalox™ C analyser following acidification and sparging. Total phosphorus (TP) was determined by the method of Eisenreich et al. (1975). Soluble reactive phosphate (SRP) and ammonium-nitrogen were determined colorimetrically.

3.8.2 Hydrochemical grouping

Water chemistry from the peat piezometers falls into three distinct groups, initially distinguished on the basis of their SEC (Figure 3-33). Group 1 comprises piezometers 14-19 targeting the temperature anomalies, where peat water chemistry is akin to the chalk waters. Group 2 includes piezometers 2 and 3 which are considered intermediate waters between Groups 1 and 2. Group 3 encompasses piezometers 1 and 4-13 where peat waters are characterised by elevated alkalinity, Ca, DOC, Si, NH₄, TP, SRP, Fe and Mn, whilst they are depleted with respect to SO₄ and NO₃. pH does not vary between groups with a mean of 7.1, 7.0 and 7.1 for Groups 1, 2 and 3, respectively. There are strong positive correlations between alkalinity, DOC and Ca in Group 3 (Figure 3-34). Group 1 and chalk waters, in contrast, are clustered, with Group 2 waters falling between.

Gravel groundwater is reasonably well-mixed across the site, although alkalinity, TP and NO₃ show some variation between groups. The gravel waters show a similarity to chalk waters. Nevertheless, in places, the gravel waters display occasional characteristics of Group 3 peat waters. This is most significant at piezometer 6 which has comparably high SEC (0.7 mScm⁻¹) and alkalinity (6.3 mEqL⁻¹), high concentrations of Ca (132.8 mgL⁻¹),

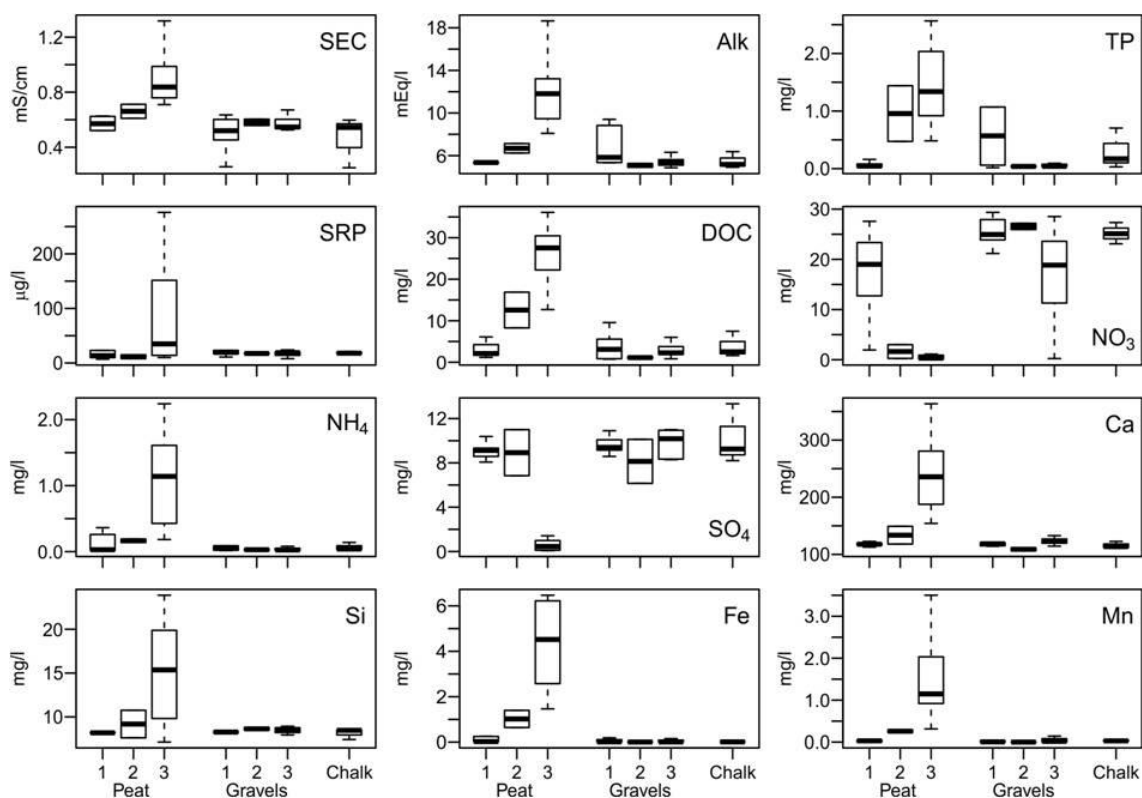


Figure 3-33. Boxplots showing selected hydrochemistry of peat, gravels, and chalk. Peat and gravels are split into three groups to reflect variations in chemistry within the peat. Chalk waters encompass both groundwater and surface waters

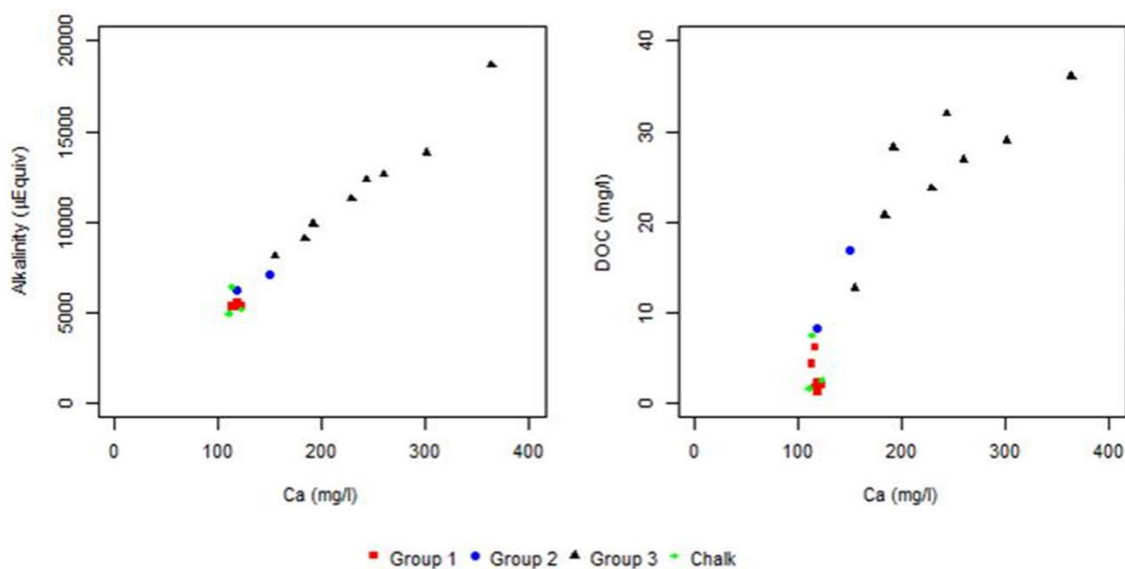


Figure 3-34. Grouped peat and chalk water concentrations of Ca against Alkalinity and DOC

DOC (5.7 mg l^{-1}), Si (8.9 mg l^{-1}), Mn (0.1 mg l^{-1}), and is depleted in SO_4 (4.3 mg l^{-1}) and NO_3 (2.6 mg l^{-1}).

3.9 Linking wetland vegetation distribution to hydrology

3.9.1 Vegetation survey

Vegetation species were identified at a subset of the temperature survey positions to identify potential botanical indicators of groundwater upwelling. The temperature survey positions were selected by stratified random sampling (De Gruijter et al., 2006) where the strata comprised deciles of the observed temperature at 0.15 m depth (Table 3-4). Twelve of the positions within each stratum were selected at random. Vegetation species were identified at these 120 locations in addition to the locations of the paired piezometers (Figure 3-35).

The survey was conducted with a 2 m² quadrat in July 2013 when stands were mature enough to allow easier identification. Each quadrat was aligned north-south and located at its southwest corner using Trimble R8™ dGPS. For each quadrat, individual species and their percentage cover were identified. These results were then used to allocate the location to a particular community of the National Vegetation Classification (NVC) (Rodwell, 1991) using the TABLEFIT procedure (Hill, 1996). This establishes the degree of agreement between species coverage in each quadrat and the association tables in British plant communities. Only where goodness of fit values were at least 50 % were samples allocated. Further, the positions of all *Carex paniculata*, easily recognisable in dense tussocks up to 1.5 m tall and 1 m in diameter, were recorded and matched to the local temperature decile.

3.9.2 Vegetation coverage

In terms of NVC plant communities the S28 *Phalaris arundinacea* tall-herb fen community dominates the site (Figure 3-36). Other prevalent communities include the OV24 *Urtica dioica* – *Galium aparine* community, OV26 *Epilobium hirsutum* community, S7 *Carex acutiformis* swamp, S6 *Carex riparia* swamp, and S5 *Glyceria maxima* swamp. There is some succession, with plant communities graded from swamp dominated by *Glyceria maxima* and *Carex acutiformis* in the north and centre to tall-herb fen dominated by *Galium aparine* and *Urtica dioica* in the south. Scattered patches of woodland (W5, W6 and W24), tall-herb fen (OV24, OV26) and fen-meadow (M28) occur to the edges of the site and centre of the north meadow. These communities are all distributed evenly across the temperature deciles (Table 3-5). Plant communities unclassified by the TABLEFIT procedure are also distributed evenly. Small samples of the communities S23 *Other water-margin vegetation*, and W6 *Alnus glutinosa* - *Urtica dioica* woodland are observed in the warmest decile.

Table 3-4. Grouping of temperature bands.

Group	1	2	3	4	5	6	7	8	9	10
Temperature	0 –	3.4 –	3.7 –	4.0 –	4.2 –	4.4 –	4.6 –	4.9 –	5.5 –	6.8 –
range (°C)	3.4	3.7	4.0	4.2	4.4	4.6	4.9	5.5	6.8	10.0

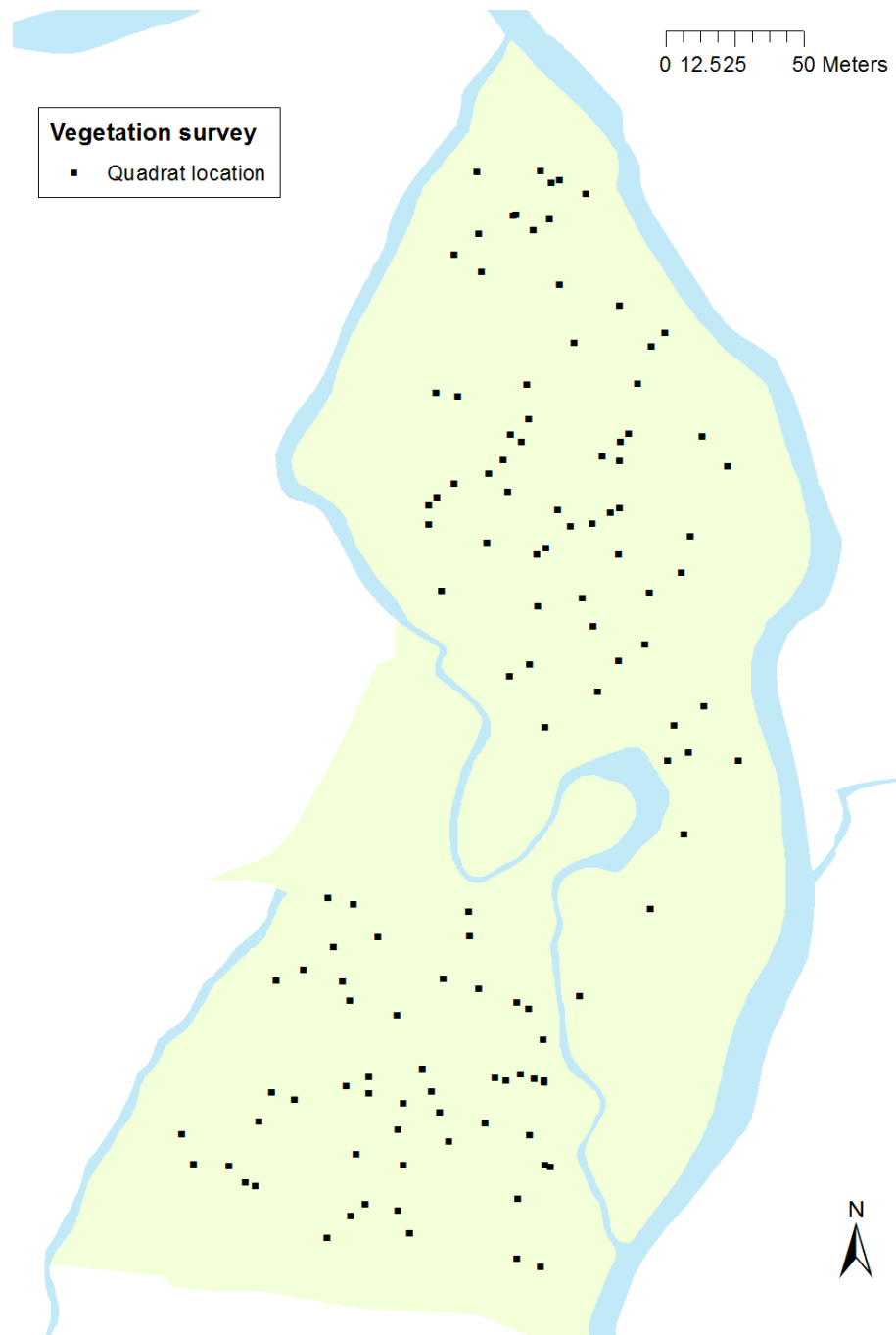


Figure 3-35. Locations of quadrats for vegetation survey.

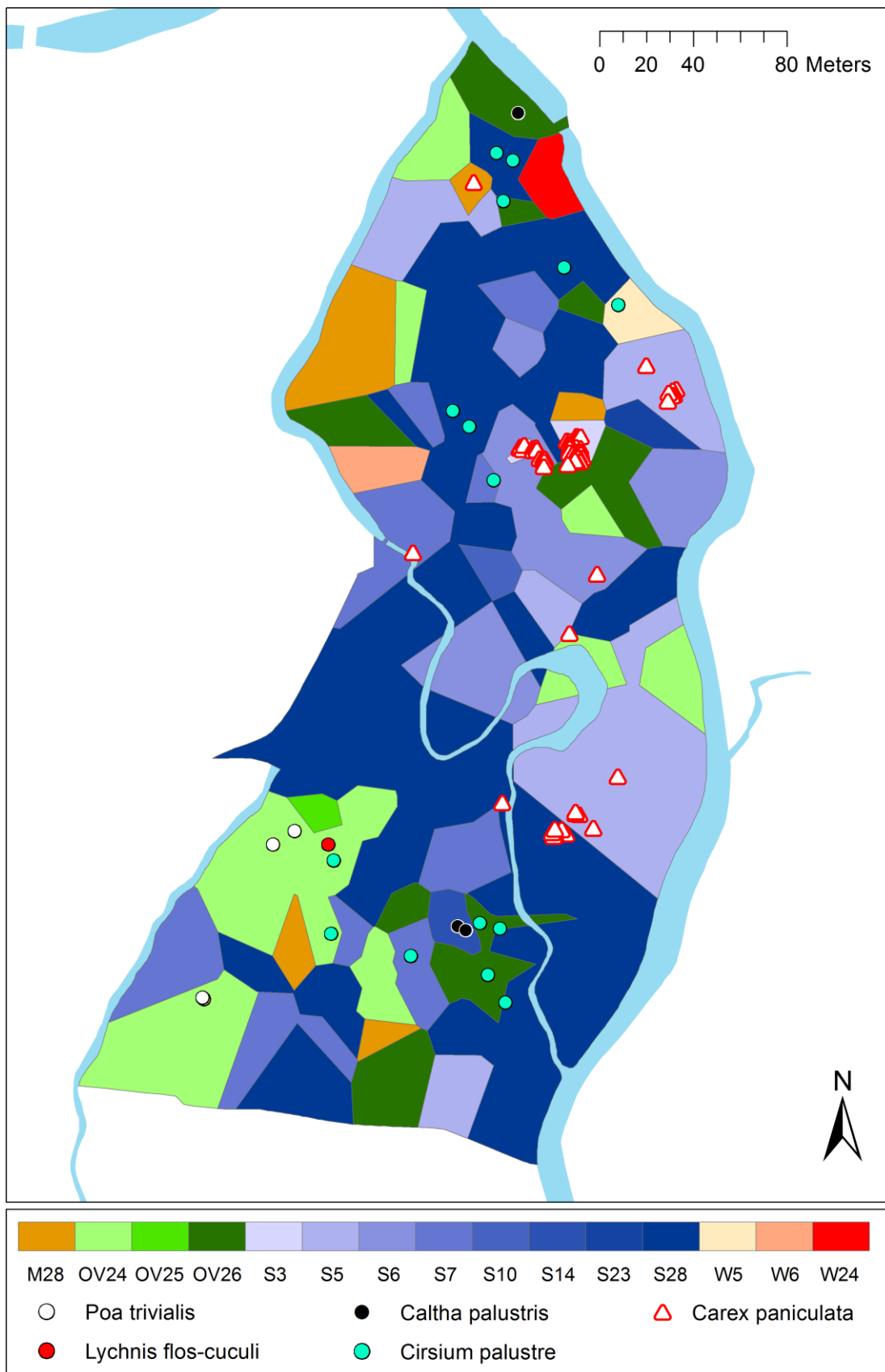


Figure 3-36. Distribution of NVC plant communities, *Carex paniculata*, and species characteristic of the MG8 community: *Poa trivialis*, *Lychnis flos-cuculi*, *Caltha palustris*, *Cirsium palustre*

Table 3-5. NVC plant community occurrences across temperature bands.

NVC community	Temperature band									
	1	2	3	4	5	6	7	8	9	10
M28		1	1		1		2		1	
OV24	4	1	4		3	3	2	1		
OV25								1		
OV26	1		4		1	1	1	2	2	3
S3						1			1	
S5	2		1	2	1	1		1	1	2
S6	2	1	1				1	2	3	2
S7	2	2	4	1		3		1	2	
S10	1									
S14			1			1				1
S23										1
S28	5	2	7	4	8	3	4	4	5	6
W24						1				
W5										1
W6										1

Amongst individual species, *Glyceria maxima*, *Carex acutiformis*, *Urtica dioica*, *Phalaris arundinacea*, *Iris pseudacorus* and *Galium aparine* prevail (Table 3-6). Scattered samples of species characteristic of the MG8 community, for which the site is designated, were mostly found in transects located in the south of the Observatory (Figure 3-36). These include *Caltha palustris*, *Poa trivialis*, *Cirsium palustre* and *Lychnis flos-cuculi*. Occurrences of *Cirsium palustre* and a single specimen of *Caltha palustris* were also found across parts of the north meadow.

The dominant species show no preference for areas with temperature anomalies. Where species appear to show an affinity to areas with warmer temperatures, such as *Salix triandra*, *Salix fragilis*, *Carex paniculata*, *Caltha palustris*, *Typha latifolia* and *Lamium album*, sample numbers are limited (1-3 locations). *Poa trivialis* appears to show the converse preference towards cooler areas, but this is based on only four samples.

The extensive *Carex paniculata* survey identified 55 of 59 individuals located in temperature deciles 9 and 10 (Figure 3-37). Furthermore, *Carex paniculata* was abundant in areas of the north meadow, but restricted to only a single individual in the south meadow (Figure 3-36).

Table 3-6. Individual species occurrence by number of quadrats across temperature bands.

Species	Temperature band									
	1	2	3	4	5	6	7	8	9	10
<i>Angelica sylvestris</i>	1	2	2	2	1		2	2		1
<i>Arrhenatherum elatius</i>		1	2							
<i>Callitriche</i> spp										1
<i>Caltha palustris</i>										3
<i>Calystegia sepium</i>	4		4	1	1		2	4	1	4
<i>Cardamine</i> spp										2
<i>Carex acutiformis</i>	8	9	11	9	8	10	11	14	7	10
<i>Carex paniculata</i>										2
<i>Carex riparia</i>	4	4	2	2		1		6	1	6
<i>Cirsium arvense</i>	6	4	8	3	5	7	6	5		
<i>Cirsium palustre</i>			1	1	1	3	1	1	2	2
<i>Cirsium vulgare</i>									1	1
<i>Epilobium hirsutum</i>	2	1	2	4	1	3	2	4	3	4
<i>Epilobium palustre</i>		1								
<i>Equisetum fluviatile</i>	5	3	2	2	1		1	3	4	5
<i>Equisetum palustre</i>	4	2	4	2	2	3	1	4	1	6
<i>Eupatorium cannabinum</i>	3		2	1	1	2	1	3	1	4
<i>Filipendula ulmaria</i>	3	6	9	7	7	7	5	6	4	4
<i>Fraxinus excelsior</i>										1
<i>Galeopsis bifida</i>			1				2		1	3
<i>Galium aparine</i>	13	9	15	9	9	8	11	10	3	3
<i>Galium palustre</i>		1				2		1	1	2
<i>Glyceria maxima</i>	10	9	13	7	7	4	9	8	5	15
<i>Hedera helix</i>										1
<i>Humulus lupulus</i>	2			1		2	3	1		
<i>Iris pseudacorus</i>	4	9	13	10	4	5	9	15	8	11
<i>Lamium album</i>								1		
<i>Lathyrus pratensis</i>			1	1		1				
<i>Lemna minor</i>								3		3
<i>Lolium</i> spp	1							1		
<i>Lycopus europaeus</i>			1							
<i>Lychnis flos-cuculi</i>	1									
<i>Mentha aquatica</i>	1	1		2				1	2	2
<i>Oenanthe crocata</i>	1			1						2
<i>Persicaria amphibia</i>			1	1		3		4	1	3
<i>Petasites hybridus</i>	7	3	5	4	2		1	4	1	1
<i>Phalaris arundinacea</i>	14	9	12	9	7	4	8	13	9	17
<i>Poa trivialis</i>	4									
<i>Rubus fruticosus</i>		1						1	1	2
<i>Rumex hydrolapathum</i>	1		1	2	1		1	2	2	
<i>Salix cinerea oleifolia</i>			1		1		1	1		
<i>Salix fragilis</i>										1
<i>Salix triandra</i>									1	2
<i>Scutellaria galericulata</i>			2					2	1	
<i>Solanum dulcamara</i>	1	1	4	1		1	1	3		6
<i>Sparganium erectum</i>	1		2					3		4
<i>Sphagnum</i> spp	3		1		1			1		
<i>Typha latifolia</i>										1
<i>Urtica dioica</i>	12	8	12	6	9	8	11	5	2	3
<i>Vicia cracca</i>		1	1	1			1			

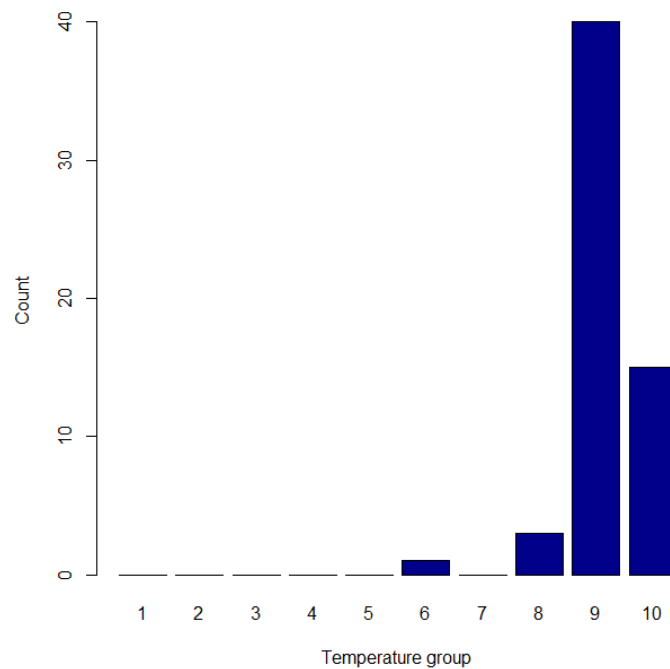


Figure 3-37. Distribution of *Carex paniculata* occurrence across temperature bands

3.10 Extent of highly weathered ‘putty’ chalk

The horizontal extent of the putty chalk was extracted from the 3D resistivity model (Chambers et al., 2014) using a representative range of resistivities (10-75 Ωm) following Crook et al. (2008). Spatial definition of the extent of putty chalk was limited by the boundaries of the ERT survey. Absence of putty chalk is seen to be confined to the north meadow (Figure 3-38). The main grouping, including a substantial feature with area of 2200 m², is situated in the centre of the north meadow. A smaller absence of this layer is located at the south of the north meadow near the duck pond (230 m²).

3.11 Discussion of groundwater/surface-water interaction

3.11.1 Groundwater upwelling and hydraulic continuity

Similarities in water level responses to rain events and macrophyte growth and its management by cutting indicates the peats, gravels and channels are in hydraulic continuity. Discrete areas of upwelling chalk groundwater are considered to be represented by warm temperature anomalies in the winter survey, mainly found in the north meadow. This is supported by groundwater head and hydrochemical data which indicate groundwater mounding, distinctive vertical head gradients, and peat water chemistry representative of chalk waters in such areas.

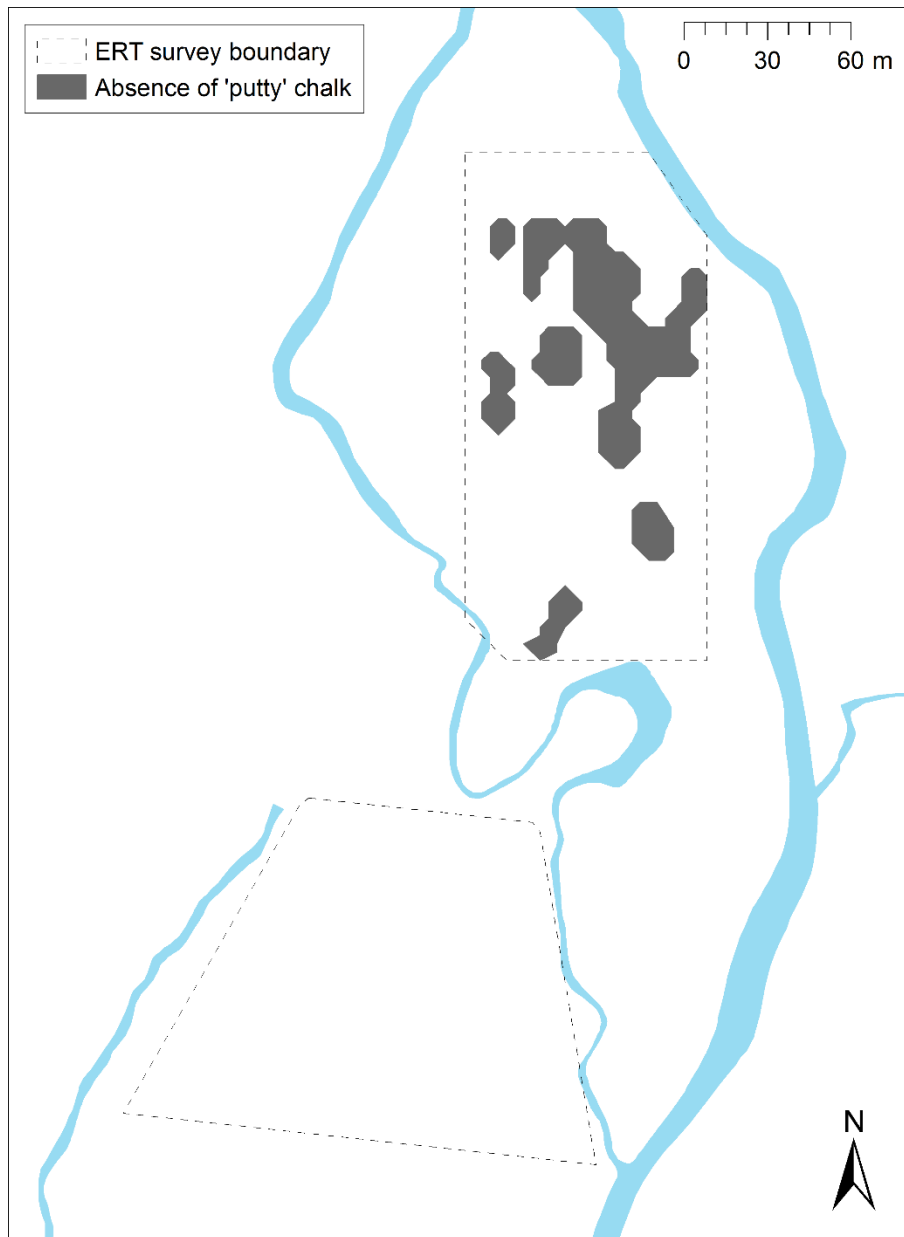


Figure 3-38. Horizontal extent of the ERT survey and absences in highly weathered 'putty' chalk

In the north meadow, peat and gravel groundwater heads are likely to be supported by upwelling groundwater from the Chalk. This is borne out by warmer temperatures in both the peat and gravels. Variations in heads around the main relic channel indicate this is likely to be the focus of upwelling. Small-scale variations in gravel head have also been noted 100 m upstream of the wetland where they were attributed to probable upwelling of chalk waters associated with geological heterogeneities (Allen et al., 2010). Groundwater mounding can be seen to occur in the northern meadow. These areas in the north meadow are concomitant with locations where the relatively impermeable putty chalk is absent at the interface between the chalk and gravel aquifers. A discrete area of

persistent and particularly high groundwater head, apparent towards the north of the site yet beyond the geophysical survey extent, could be likewise accounted for by an absence of putty chalk.

The warm temperature anomaly within the sinuous relic channel of the north meadow is maintained at depth, yet loses constancy towards the surface. This is plausibly a result of groundwater discharge of varying magnitudes. However, during hydrochemical sampling when the peat was saturated across the site, piezometers away from the channel flowed intermittently when abstracted. Peat piezometers within the channel, however, sustained continuous abstraction indicating a higher permeability. Hence, localised upwellings and resultant increases in hydraulic head would promote preferential lateral flow through more permeable sediments of the relic channel. The observed heat transport could then encompass an element of lateral advection induced by relic channel flow.

Hydrochemical evidence suggests some lateral groundwater movement away from the vicinity of relic channels. This is supported by the intermediate group 2 waters contained within peat piezometers 2 and 3. Peat piezometer 3 is located down gradient of the northern relic channel, from which water is likely to have at least partially originated given the lack of upward hydraulic gradient at the site itself. A relic channel exists 5-8 m southwest of piezometer 2, yet beyond the survey extent.

At a distance of approximately 370 m along the relic channel temperature anomalies cease. Further south there is a large inversion in the vertical head gradient indicating the potential for downward movement of peat waters into the gravels. Downwelling of peat waters within this vicinity is further supported by hydrochemical and temperature data at gravel piezometer 6.

In the south meadow, the main warm anomaly is contained within a minor relic channel within 30 m of the Westbrook. This anomaly is relatively cooler compared to those in the north meadow indicating less significant groundwater discharge. Moreover, there is generally no significant gradient between the groundwater and surface waters. Therefore groundwater is not considered to be a major control on peat heads in the south meadow. Heads are likely to be supported by river stage and/or rainfall.

3.11.2 Biogeochemical wetland processes

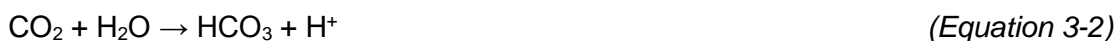
The wetland appears to be acting as a highly dynamic biogeochemical reactor (Prior and Johnes, 2002). Upwelling areas are delivering waters rich in NO_3 and SO_4 , which are removed away from the location of the temperature anomalies through reductive

bacterial processes. On the other hand, the upwelling waters are depleted with respect to SRP and TP. Nevertheless, SRP concentrations ($\bar{x} = 53.1 \mu\text{gl}^{-1}$, $\sigma = 76.4 \mu\text{gl}^{-1}$) suggest that P is not a limiting nutrient for the wetland ecosystem. These biogeochemical processes and nutrient fluxes create distinct chemical environments, which have implications for ecological response.

The reductive processes and chemical observations can be explained through equations 1 to 6 (Gooddy et al., 2002). High alkalinity and reducing conditions away from the upwelling waters may be explained in part through the oxidation of organic matter:



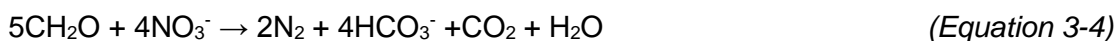
This dissociates to:



The free proton is then able to liberate calcium from the calcite in the peaty matrix:



The much lower NO_3^- concentrations away from the upwelling waters provide some evidence for denitrification:



and nitrate reduction to ammonia (DNRA) with higher NH_4^+ and DOC concentrations:



which both further contribute to the alkalinity. Although a number of studies have supported the premise that high organic carbon availability favours DNRA over denitrification (Megonigal et al., 2003), the higher proportion of NH_4^+ is likely due to mineralisation of organic N in the peat. Denitrification could also produce N_2O as the final product.

Similarly, lower SO_4^{2-} concentrations could be from bacterial sulphate reduction:



which, again, add to an increase in the alkalinity. Similar sequences of redox chemistry have been observed in other calcium carbonate environments when stimulated by high loadings of organic carbon (Gooddy et al., 2002).

3.11.3 Botanical indicators of groundwater discharge

The NVC plant communities identified for the site are species poor and collectively typical of mesotrophic to eutrophic soils and waters (Wheeler et al., 2004). There is little discernible preference to water source as indicated by temperature. Some individual species seem to correspond to areas of groundwater discharge in the main vegetation survey, but sample numbers are low. These include *Caltha palustris* which has been recognised elsewhere as a possible indicator of groundwater discharge (Klijn and Witte, 1999; Rosenberry et al., 2000; Wierda et al., 1997).

The affiliation of *Carex paniculata* to warm temperature anomalies is marked. An explicit association between the presence of the species and temperature is, however, doubtful. *Carex paniculata* is part of a widespread European temperate element in the UK flora, as defined by Preston and Hill (1997). There is little suggestion that the species has a tight temperature requirement, though Ellenberg (1988) has it as an “indicator of fairly warm conditions from lowland to high mountain sites, but especially in submontane to temperate regions”. The occurrence of this sedge is more likely to be determined by other factors linked to the temperature pattern.

The upwelling areas deliver waters relatively low in minerals Ca, Si, Fe and Mn, and nutrients TP and SRP, yet rich in NO₃ and SO₄ (Figure 9). A preference for *Carex paniculata* to Ca poor groundwater with higher NO₃ and SO₄ concentrations has been shown on the Pleistocene sands of central and northeast Netherlands (Grootjans et al., 1988; Wassen et al., 1988). Furthermore, *Carex paniculata* has been shown to prefer waters with low Ca and relatively high NH₄ concentrations on the gravel deposits of the River Meuse in southeast Netherlands (Lucassen et al., 2006). However, groundwater Ca concentrations are higher across the study site, given its setting within a chalk valley, than those found in the Netherlands, by around 30-70 mg l⁻¹, as are Si concentrations by 3 mg l⁻¹. Fe and Mn concentrations are comparatively lower, by around 1-12 mg l⁻¹ and 1.7 mg l⁻¹, respectively.

At this site, it is considered that nutrient rather than mineral supply is likely to be the limiting factor for *Carex paniculata*, particularly given it is a sizeable species with high nutrient demands. Furthermore, it is plausible that NO₃, rather than P, is the limiting nutrient for *Carex paniculata* distribution. Addition of N to calcareous fens has been shown to increase the biomass of some *Carex* species, without changing the species composition (Pauli et al., 2002). Increased P has also been found to promote the growth of poor fen species, especially more competitive grasses, to the exclusion of other species (Hoek et al., 2004). These findings are consistent with evidence of *Carex*

paniculata surrounded by a variety of poor fen species to situations with the influx of base-rich groundwater and eutrophication (Rodwell, 1991; Sinker, 1962). This suggests the species may have potential as an indicator of groundwater discharge across lowland chalk wetlands, as chalk groundwater is generally high in NO_3 throughout much of Western Europe due to historical agricultural loadings (Aguilar et al., 2007; Wang et al., 2012).

The value of *Carex paniculata* as an indicator species lies in its use for initial hydrological site appraisals and directing further study. The presence of the species or of groundwater discharge does not guarantee the other, despite the majority of stands found in the groundwater dependent north meadow. Rather, tussocks of *Carex paniculata* indicate adjacent conditions arising from a distinct chemical environment. These may result from groundwater discharge in a particular geological context or from another supply mechanism (Wheeler, 1999). Furthermore, other factors than water source are influential, principally light availability (Goslee et al., 1997), along with soil conditions, air quality and seed availability. Supporting information is invariably required. Wider areas may be characterised as potential areas of groundwater discharge, although it is not possible to delineate discrete areas by vegetation alone.

3.11.4 Value and limitations of a high resolution 3D temperature model

A detailed 3D temperature model could be a useful precursor to the targeted deployment of sensor arrays. One-dimensional vertical temperature arrays are becoming increasingly commonplace as a means of estimating groundwater fluxes (Anibas et al., 2009; Voytek et al., 2013). Such estimates require profile time series to solve analytical flux representations (Briggs et al., 2012; Hatch et al., 2006). Sensors could be positioned to sample temperature gradients representatively across an entire site to estimate total groundwater influx. Furthermore, their deployment could also be based upon an understanding of the flow field, which is important to avoid misinterpretation of temperature time series (Cuthbert and Mackay, 2013). For example, at this site there is evidence for non-vertical flows which have been considered the greatest source for error when implementing 1D solutions (Lautz, 2010).

Temperature data may be gathered simply and economically over large areas. In this study the total equipment cost was GBP 250 and a team of two covered 0.1 ha hour^{-1} . Limitations of the technique include the need for a high water table elevation and penetrable soils, although these are often distinguishing features of wetlands. Furthermore, it is possible that groundwater discharge may exhibit seasonality or

dynamically respond to intense precipitation events, which would not be captured in a single temperature survey.

Other approaches to detect groundwater upwelling include water budgeting (Acreman and Miller, 2006), intrusive investigation, and remote sensing by thermal or multispectral imaging (Becker, 2006). Water budgeting lacks spatial definition and requires quantifiable boundary flows which are often not applicable to wetlands. Standard intrusive exploratory techniques include drilling and trial pitting which are disruptive to sensitive ecosystems and are spatially restricted (Baines et al., 2002). The influence of air temperature at the ground surface renders remote sensing uncertain, as groundwater temperature signals are seen to fade noticeably in the upper 0.5 m. Moreover, commonly used satellite techniques do not possess the requisite spatial resolution and have limited penetration of the subsurface (Becker, 2006). The method presented is able to encapsulate the necessary scale, depth and resolution with minimal intrusive impact.

3.12 Summary and conceptual model

An extensive field campaign has built upon previous investigations, particularly a 3D ERT survey, to enable the development of a conceptual model of the hydrological functioning of the Observatory. The relatively flat site, sloping gently to the south, follows the line of the Lambourn valley. Generally, head contours in the peat, gravels and chalk also follow this direction. A network of shallow, infilled relic channels traverses the site and connect to the main River Lambourn and Westbrook channels and a narrow drainage channel in the southwest of the site. Flows in the main channels display a non-flashy regime with continuous baseflow and runoff dominated floods. The fluvial regime is dominated by in-channel macrophyte growth and its management by periodic cutting. These have an acute effect on water levels, which otherwise respond to seasonal and event changes in precipitation and evapotranspiration, resulting in a non-curvilinear stage-discharge relationship.

The peat and gravels are considered to have good hydraulic connectivity, with head boundaries in the River Lambourn and Westbrook broadly controlling water levels across the wetland. A double aquifer system of gravels and Chalk is mostly separated by a confining layer of low permeability 'putty' chalk (Figure 3-39). Leakage occurs between the gravels and Chalk where the putty chalk is absent, causing localised variations in water levels in areas that had not previously been considered to be groundwater

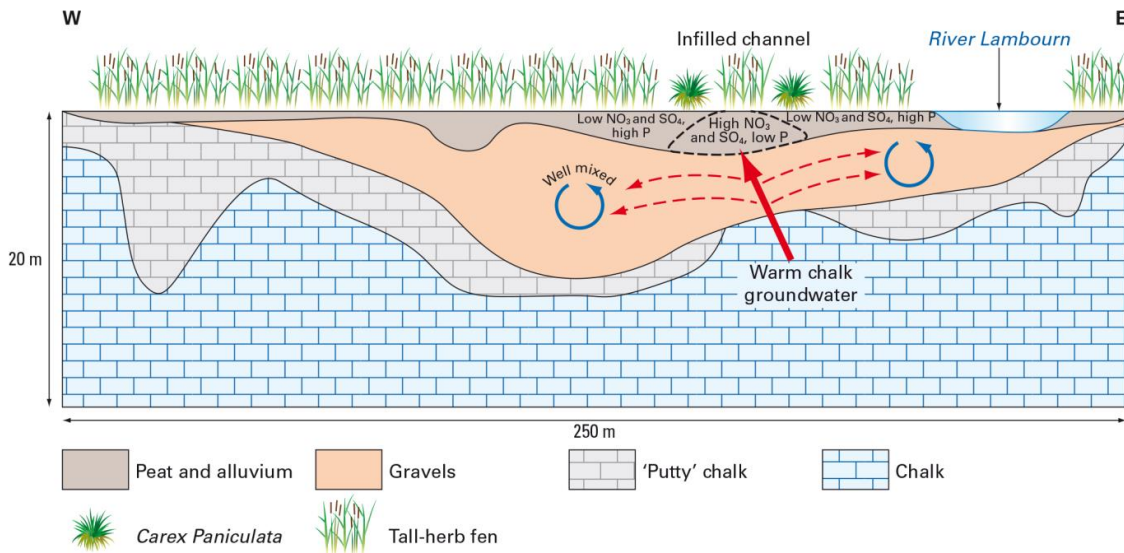


Figure 3-39. Conceptual vertical section through the Observatory

dependent. These are concurrent with the relic channels in the peat and occur predominantly in the north meadow. In the south meadow groundwater is not considered to contribute to wetland water levels, which are controlled by river stage and/or rainfall. Upwelling chalk groundwater contains high concentrations of nitrate which is considered to support the spatially restricted growth of *Carex paniculata* against a background of poor fen communities located in reducing higher-phosphate waters. Remnants of the MG8 community, for which the site is designated, occur largely in the south.

The conceptual model allows identification of the hydrological processes and interactions that contribute to the site's functioning. These provide the basis for simulation with a hydrological model, which may be used to improve understanding and predict the hydrological effects of environmental changes, such as through abstraction, climate change or modifications to the management regime. Selection of a suitable model, its development and use in understanding the hydrological and ecological system form the focus of the next chapters.

Chapter 4

Hydrological modelling of the River Lambourn Observatory

4.1 Introduction

Conceptual understanding of the Lambourn Observatory leads to identification of applicable hydrological processes and their possible interactions. Simulation of these processes requires identification of an appropriate model, or models, and considerations of model architecture and connections. The selection of a suitable hydrological modelling system for the Observatory forms the focus for the first half of this chapter. Many commercially and academically available models were excluded due to the need for adequate representation of groundwater/surface water interaction. The MIKE SHE modelling system was chosen out of those which met selection criteria developed from the conceptual model detailed in the previous chapter. Development of the hydrological model of the Observatory, its calibration and validation against observed values, and the model performance against statistical measures form the focus for the second half of this chapter. The numerical model was used to quantify the water balance and enhance understanding of the site's hydrological functioning, with particular regard to surface water flooding and upwelling groundwater, and groundwater/surface water interaction.

4.2 Requisites for modelling hydrology at the River Lambourn Observatory

A crucial need to represent interaction between surface and groundwater is evident from the conceptual model. Such interaction is complex, and physically dependent on topographical, geological and climatic factors (Sophocleous, 2002; Winter, 1999). Due to this level of complexity, hydrological models have generally been designed either for surface water processes or groundwater flow explicitly. Research to simulate groundwater/surface water interactions has centred on coupling surface and subsurface flows at the interface through the internal boundary condition of infiltration (Morita and Yen, 2000). Some examples include Meyboom (1961), Pinder and Jones (1969), Freeze (1972b), Cunningham and Sinclair (1979), Abbott et al. (1986a), Abbott et al. (1986b), Panday and Huyakorn (2004), Markstrom et al. (2008).

Three methods commonly exist for conjunctive simulation of surface groundwater flows (Morita and Yen, 2000). The simplest, yet least accurate, is to solve the uncoupled surface and subsurface models separately and successively without iteration (Smith and Woolhiser, 1971). An infiltration rate expression is used as the boundary condition. A

relatively short runoff period and quick change in flow generally means the surface flow model is solved initially. Values of infiltration are then passed to the subsurface flow model at the equivalent time step before moving on to the next. Secondly, surface and subsurface domains may be coupled by solving separately yet iteratively at the same time step. A gradient type equation represents infiltration as the internal boundary condition (Freeze, 1972a; Pinder and Sauer, 1971). Iterative errors at a certain time step must be within prescribed tolerances before progressing to the next. The third method is to solve the surface flow equation, subsurface flow equation, and internal boundary condition infiltration equation simultaneously with each successive time step (Morita and Yen, 2000). There is a relative increase in complexity with each method, with the third being the most sophisticated and, theoretically, satisfactory.

Although models may be lumped, semi-distributed, and include elements of both, this study focuses on fully distributed discretisation due to advantages in representing spatial variability. Spatial domains may be distinguished between the channel, overland flow, and subsurface flow. Overland flow and subsurface domains may be discretised using finite difference or finite element methods (Figure 4-1). Domains are divided into a 2D or 3D grid mesh for overland flow and subsurface flow respectively. The finite difference method computes values for nodal points set either at the intersections of a rectangular grid mesh or at the centre of each grid cell or block (Jones, 1997). The finite element method calculates values at nodes as well as interpolating values between nodes. It is able to incorporate other mesh configurations than a rectangular grid.

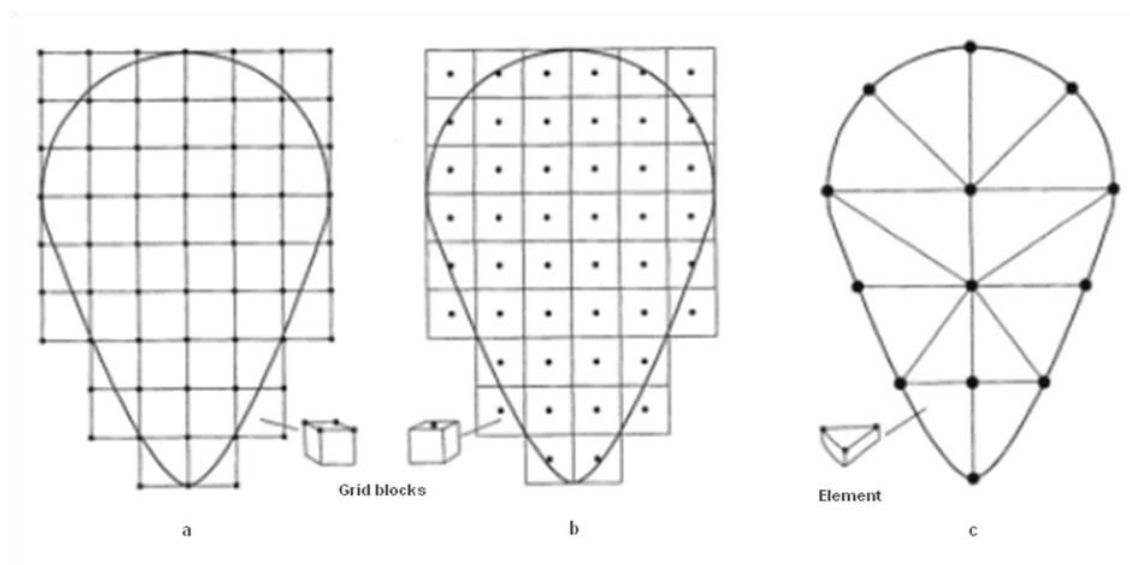


Figure 4-1. Finite difference and finite element representations of a catchment, after Jones (1997): a) finite difference with nodes at grid points; b) finite difference with nodes at cell centres; c) finite element

The channel network may be discretised using a finite volume approach. A set of 1D line elements represents interconnected reaches, with nodes at their centres. One or more channel segments may be connected to a single node of the overland flow or subsurface domains, or may span several nodes. To join nodes of each domain requires either numerical superposition, where the top layer of nodes represents both surface and subsurface flow or flux terms, to transfer values between each layer.

The type of equations, spatial dimension, and coupling method between surface and groundwater components have led to the development of models at different levels of complexity (Saleh et al., 2011). With greater complexity such models become increasingly restricted due to problems of (i) extreme data demand and inability to calibrate outputs, (ii) translation of physical equations to large spatial scales, and (iii) computational power (Beven and Binley, 1992). To manage issues of computational power and scale, Aral and Gunduz (2003) suggested that an approximate formulation of coupled surface and groundwater flow may be achieved through use of a multi-layer 2D horizontal model for subsurface flow, preceding a 1D model for surface flow. However, ultimate decision rests on the research needs and existing resource availability. Evaluation of these needs led to the identification of criteria to aid in selection of a suitable model (Table 4-1).

4.2.1 Comparison of hydrological modelling systems

The fundamental requirement to adequately simulate groundwater/surface water interaction precludes many models from consideration. The academically and commercially prominent models chosen for review in this study comprise HGS, MIKE SHE, MODHMS, SHETRAN, and WaSIM. These are, necessarily, physically-based, distributed and either fully integrated or coupled. Categorising model suitability rests on functionality within these principal characteristics, along with assessment of the other remaining selection criteria.

Criteria were categorised with low, medium and high ratings to support the ultimate choice of model. Table 4-2 details the procedure for rating criteria against the level of provision for each model.

Selection criteria was also aggregated into three evaluative groups allowing further assessment of model capability against requirements:

Table 4-1. Criteria for model selection

Selection criterion	Rationale
Physically-based	Due to the complexity of hydrological processes within the wetland, calibration should be minimised. A physically-based model will provide a relatively accurate representation of such processes, whilst parameters can be determined and evaluated using measurable physical quantities.
Fully distributed	The model should be able to predict hydrological response at any point of interest within the study area.
Surface-groundwater interaction	A significant groundwater flow component is essential, and should be effectively coupled to the surface water component. Interaction between the wetland, groundwater and channel flow is fundamental to defining the dominant hydrological processes.
User friendly	A modular structure enables individual parts of the model to be altered, without affecting the whole. Particular processes may then be changed simply and efficiently. An intuitive GUI should be incorporated, along with straightforward synthesis with GIS applications.
Wetland applicability	Proven application to wetland hydrological modelling.
Automatic calibration	The need for calibration is an inevitable procedure, yet may require excessive time if carried out manually.
Temporal representation	Simulation should be continuous to include unsteady flow in storm events as well as low-flow conditions during dry periods. A variable time step is needed in order to save computation time in a continuous model run. Sub-hourly increments may be applied during storm events, and longer time steps for dry periods.
Suitability of scale	The model should be able to be applied at a range of scales from a soil column or transect, to the full watershed, in order to focus on different facets of water flux in the wetland.
Support and training	Existing expertise in model use should be readily available. Established training programmes and documentation should facilitate succinct understanding.

- Functionality - physical representation, distribution, integration
- Adaptability - calibration, temporal representation, scale
- Accessibility - ease of use, applicability, training

A general description of each model follows with tabulated summaries of model capability against each selection criterion.

Table 4-2. Criteria rating for model capability

Criterion	Model capability	Rating
Physical representation	One or more processes not represented	L
	All processes represented but choice is non-existent or limited	M
	All processes represented with choice of method available	H
Distribution	Semi-distributed	L
	Finite difference	M
	Finite element	H
Integration	Surface and subsurface flow equations solved separately with infiltration rate term as boundary condition	L
	Surface and subsurface flow equations solved separately yet iteratively at the same time step	M
	Surface and subsurface flow equations solved simultaneously for each time step	H
Ease of use	No GUI, non-modular, no link to GIS	L
	Some GUI, modular, or GIS capability	M
	Bespoke GUI, modular, and GIS capable	H
Applicability	No wetland studies applications	L
	Between 1 and 3 wetland applications	M
	Over 3 wetland applications	H
Calibration	Manual or external programme	L
	Autocalibration	M
	Autocalibration and sensitivity analysis	H
Temporal representation	All processes solved at a uniform time step	L
	All processes solved at a variable time step	M
	Each process solved at a relevant time step	H
Scale	Application to one scale range	L
	Application to a partial range of scales	M
	Application to full range of scales, from soil column to watershed	H
Support and training	No training or support	L
	Some training or support	M
	Full training and support	H

HGS

HGS (HydroGeoSphere) is a fully coupled, physically-based, spatially distributed, integrated surface water and groundwater model (Therrien et al., 2010). It was developed by the University of Waterloo and Université Laval from the 3D subsurface flow and transport code, FRAC3DVS. Both surface and subsurface domains are discretised using a finite element approach, allowing full distribution in both horizontal and vertical directions (Table 4-3). A flexible time-stepping algorithm is used to reduce simulation run

Table 4-3. HGS model capability against rating criteria

Criterion	Model capability
Physical representation	Interception: Maximum storage threshold after Wigmosta et al. (1994) Evapotranspiration: Kristensen and Jensen (1975) Snowmelt: None Overland flow: 2D St Venant diffusive wave Unsaturated zone flow: 3D variably saturated Richards equation Groundwater flow: 3D variably saturated Richards equation Channel flow: 1D general equation of continuity of flow
Distribution	Full horizontal and vertical distribution through finite element discretisation.
Integration	Surface and subsurface flow equations solved simultaneously for each time step.
Ease of use	Text based files. Modular code. No bespoke GUI but may use FRAC3DVS tools for grid generation and subsurface flow model input, and Tecplot for visualisation of outputs. Import of spatial data with GIS (ArcGIS).
Applicability	Frei et al. (2010)
Calibration	Manual or external programme.
Temporal representation	Variable and adaptive time stepping dependent on transient behaviour of system.
Scale	Grid size centimetres to kilometres. Application to soil columns, research sites and catchments up to 1000km ² .
Support	Source code available under academic distribution and agreement. Manual provided with worked examples. No tutorial and limited model support via developer email.

times. HGS can simulate subsurface flow processes in a physically based manner, including porous media, fractures, subsurface conduits, macropores, and perched water tables. Input data demands are relatively low, with precipitation required for event simulations and evapotranspiration for continuous simulations. Although geared towards the study of surface groundwater interactions, HGS is not set up for integration in a GIS environment, and has no GUI available.

MIKE SHE

MIKE SHE is an integrated system which simulates the land-based phase of the hydrological cycle (Graham and Butts, 2005) (Figure 4-2). Developed originally from the *Système Hydrologique Européen* (SHE) (Abbott et al., 1986a; Abbott et al., 1986b) by the Danish Hydraulic Institute (DHI), it has been utilised for international river basins (Andersen et al., 2001; Stisen et al., 2008; Thompson et al., 2014; Thompson et al., 2013), catchments with areas of hundreds to thousands of km² (Feyen et al., 2000; Huang et al., 2010; Singh et al., 2010; Singh et al., 2011), to small (<50 km²) catchments and has proven applicability to individual wetlands (Al-Khudhairy et al., 1999; Thompson, 2012; Thompson et al., 2004). Process modules are included for precipitation, evapotranspiration, overland flow, unsaturated flow, groundwater flow, and channel flow, along with their interactions (Table 4-4). Channel flow is represented in MIKE SHE by the hydraulic modelling system MIKE 11, which simulates unsteady flow and water levels through a finite difference approach or simple flow routing method. MIKE SHE and MIKE 11 are coupled by solving separately yet iteratively at the same time step. Although often

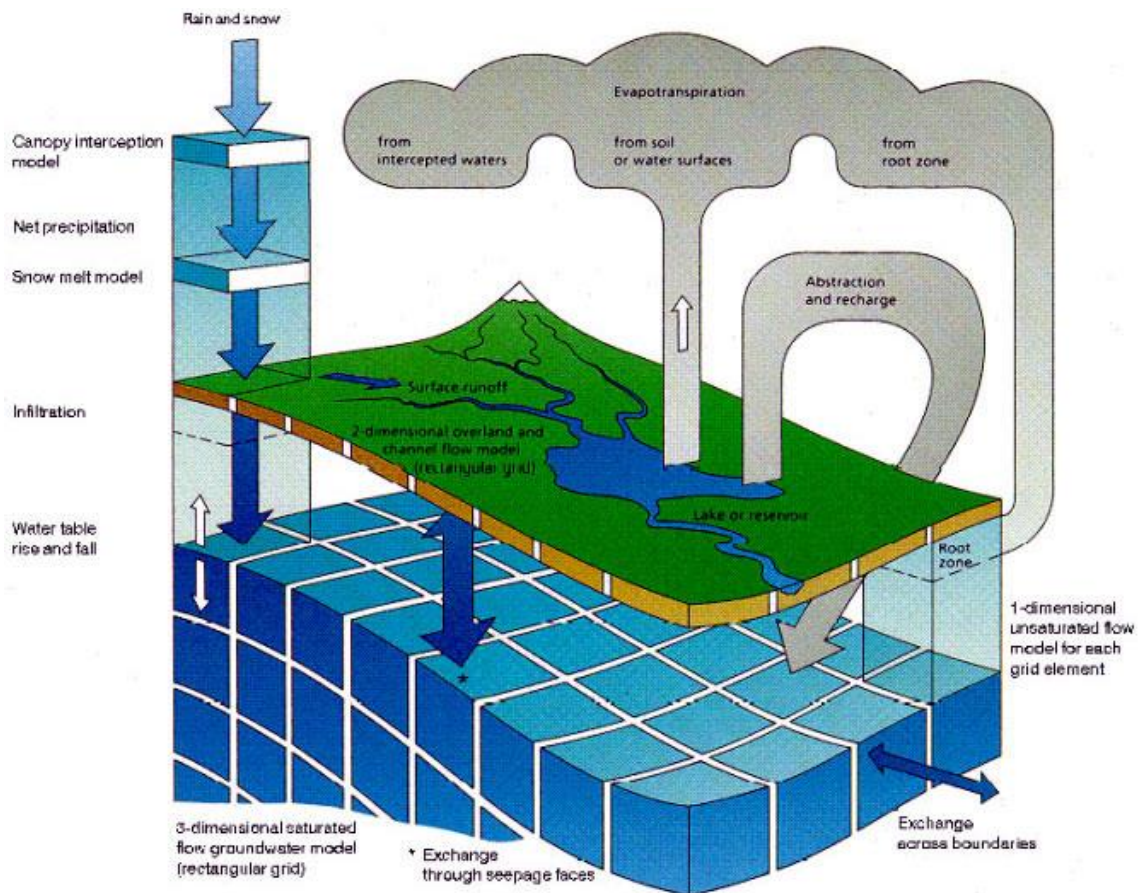


Figure 4-2. Hydrological processes simulated by MIKE SHE (DHI, 2009)

Table 4-4. MIKE SHE model capability against rating criteria

Criterion	Model capability
Physical representation	<p>Interception: Maximum storage threshold</p> <p>Evapotranspiration: SVAT, Kristensen and Jensen (1975), 2-layer water balance, or net recharge (e.g. DAISY).</p> <p>Snowmelt: Degree-day melting</p> <p>Overland flow: 2D finite difference diffusive wave, or semi-distributed.</p> <p>Unsaturated zone flow: 1D finite difference Richards equation or gravity flow, 2-layer water balance, or net recharge (e.g. DAISY).</p> <p>Groundwater flow: 3D finite difference Darcy flow, or lumped conceptual linear reservoir.</p> <p>Channel flow (MIKE 11): 1D St Venant equations – kinematic wave approximation, diffusive wave approximation, fully dynamic, higher-order fully dynamic. Flow routing – no routing, Muskingham, Muskingham-Cunge.</p>
Distribution	Fully distributed, finite difference, square grid cells only.
Integration	Surface and subsurface domains solved separately yet iteratively at the same time step.
Ease of use	<p>Modular system allows model functions to be selected based on needs.</p> <p>User friendly GUI for pre- and post-processing. Animation of model scenario results.</p> <p>Allows data import and export with GIS (ArcGIS and MapInfo) and CAD.</p> <p>Contains inbuilt spatial data editors.</p>
Applicability	<p>Refsgaard et al. (1998)</p> <p>Jacobsen et al. (1999)</p> <p>Yan et al. (1999)</p> <p>Thompson et al. (2004)</p> <p>Thompson et al. (2009)</p> <p>Staes et al. (2009)</p> <p>Duranel (2015)</p>
Calibration	<p>Contains protocol for autocalibration, sensitivity analysis and scenario management, AUTOCAL.</p> <p>Otherwise manual calibration.</p>
Temporal representation	Process-based framework allows each process to be solved at a relevant time step.
Scale	<p>Each process may be solved at relevant spatial scale.</p> <p>Grid sizes from centimetres to kilometres. Uniform grid sizes.</p> <p>Application to soil columns, research sites and watersheds.</p>
Support	<p>Comprehensive manual, tutorial and training courses easily available</p> <p>International support from software@dhigroup.com..</p>

labelled as a deterministic, fully distributed and physically based model, the complexity of process representation may be varied to include empirical and semi-distributed methods. The model may thus be built iteratively in line with data availability and process understanding.

MODHMS

MODHMS (MODular Hydrologic Modelling System) is a physically-based, spatially distributed hydrological modelling concept. It was developed by HydroGeoLogic Inc. from an enhanced variably saturated flow version, MODFLOW-SURFACT, of the USGS groundwater flow code MODFLOW (HydroGeoLogic, 2012). MODHMS is distinguished by its development objective to provide a fully integrated simulator of groundwater flow, channel flow and overland flow with dynamic coupling between each of the processes (Werner et al., 2006) (Table 4-5). The programme structure is modular, with different packages to describe each hydrological component. Derivation of equations, discretisation, boundary condition definitions and numerical solution techniques are fully discussed by Panday and Huyakorn (2004). Channel flow is represented using the CHF1 package, which is implicitly coupled with the equation for variably saturated flow through stream leakage. The groundwater and surface water conditions are solved simultaneously at a uniform time step, which may lead to intensive computational requirements for models of larger watersheds. Barr and Barron (2009) encountered

Table 4-5. MODHMS model capability against rating criteria

Criterion	Model capability
Physical representation	Interception: Maximum storage threshold Evapotranspiration: Kristensen and Jensen (1975) Snowmelt: May be specified as a flux rate Overland flow: 2D diffusive wave Unsaturated zone flow: 3D variably saturated Richards equation Groundwater flow: 3D variably saturated Richards equation, reduces to Darcy flow when fully saturated Channel flow: 1D diffusive wave
Distribution	Fully distributed, finite difference.
Integration	Surface and subsurface flow equations solved simultaneously for each time step.
Ease of use	Modular system allows model functions to be selected based on needs. Model comes with bespoke GUI. Ability to incorporate MODFLOW GUIs. Import of spatial data with GIS (ArcGIS)
Applicability	No documented wetland applications.
Calibration	Manual or external programme.
Temporal representation	All hydrologic processes solved implicitly at a uniform time step.
Scale	Grid size from centimetres to kilometres.
Support	Manual, tutorial and training courses available. Support provided from info@hglsoftware.com .

difficulties with temporal discretisation when simulating the Southern River catchment, Australia. Excessive computational resources were required to undertake sub-hourly rainfall-runoff aspects of the simulation, leading to an underestimation of channel peak flows.

SHETRAN

SHETRAN is a 3D physically-based, spatially distributed, finite difference, coupled surface/subsurface model for flow and transport (Ewen et al., 2000) (Figure 4-3). As with MIKE SHE, the point of origin for SHETRAN was the *Système Hydrologique Européen* (SHE) (Abbott et al., 1986a; Abbott et al., 1986b). In the mid-1990s, Wicks and Bathurst (1996) added a sediment transport capability to form SHESED, which was later renamed to SHETRAN, since developed within the Water Resource Systems Research Laboratory, School of Civil Engineering and Geosciences, University of Newcastle upon Tyne. The model structure is made up of three components: water flow, sediment transport, and solute transport. Flow is assumed to be unaffected by transport and sediment transport unaffected by solute transport, thus the components form a hierarchy. The land phase of the hydrological cycle is simulated in a fully integrated way for

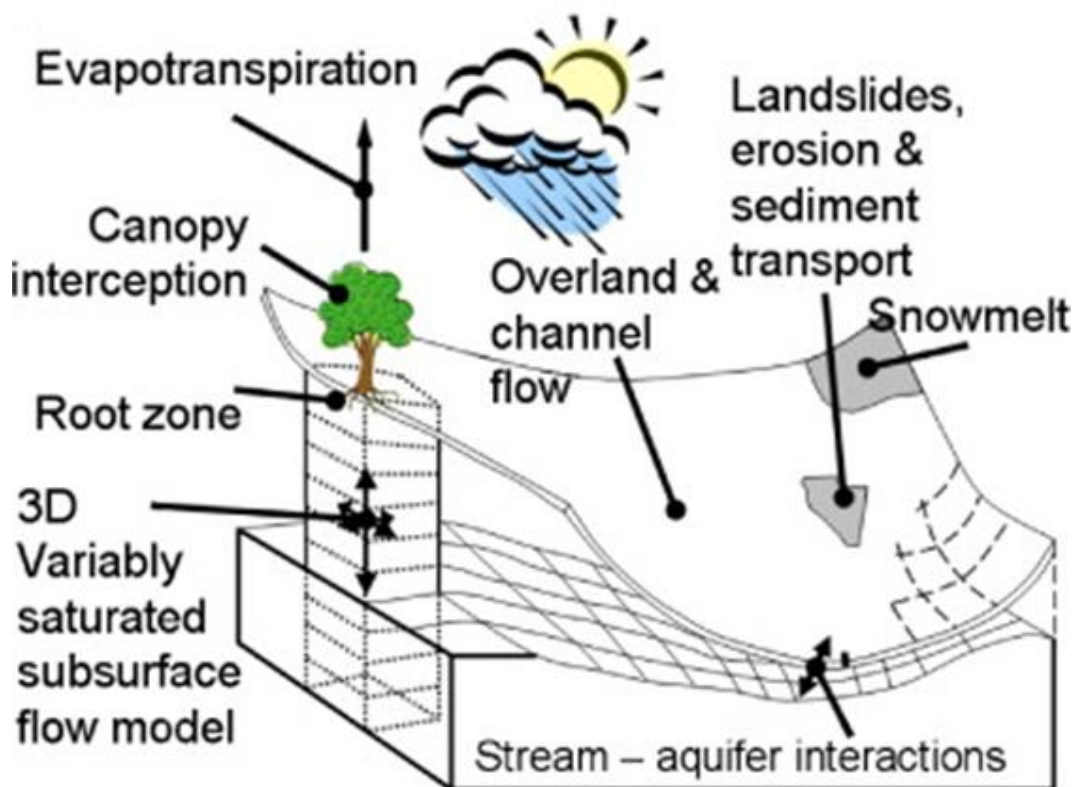


Figure 4-3. Schematic diagram of the SHETRAN model, after Bathurst et al. (2010)

vegetation interception and transpiration, snowmelt, overland flow, variably saturated subsurface flow and river/aquifer interaction (Bathurst et al., 2010) (Table 4-6). Two versions are available, one with a GUI for a Windows environment, and the standard version which uses text based files. Simulations are limited to a 50 by 50 unit grid size, unless collaborative working is agreed with the developer.

WaSIM

WaSIM (Wasserhaushalts-Simulations-Modell) is a distributed, deterministic, mainly physically based hydrological model (Schulla, 2012) (Figure 4-4). The model was initially

Table 4-6. SHETRAN model capability against rating criteria

Criterion	Model capability
Physical representation	Interception: Rutter model Evapotranspiration: Penman-Monteith (Monteith, 1965) or lumped soil moisture control analogue Snowmelt: Degree-day melting or energy budget method Overland flow: 2D St Venant diffusive wave Unsaturated zone flow: 3D variably saturated Richards equation Groundwater flow: 3D variably saturated Richards equation Channel flow: 1D St Venant kinematic wave
Distribution	Fully distributed, finite difference.
Integration	Surface and subsurface domains solved separately yet iteratively at the same time step.
Ease of use	Modular structure. GUI available for Windows environment. Otherwise text files. No link to GIS.
Applicability	No documented wetland applications.
Calibration	Manual
Temporal representation	All processes solved simultaneously. Time step may vary through simulation.
Scale	Limited to 50 by 50 units unless in collaboration with developer. Grid sizes from centimetres to kilometres. Uniform grid sizes. Application to soil columns, research sites and watersheds.
Support	Source code available. Manual and tutorial available. Online user group. Model support from developers.

developed by Jörg Schulla in 1997 under the auspices of the Swiss Federal Institute of Technology, Eidgenössische Technische Hochschule (ETH), as WaSIM-ETH (Schulla, 1997). However, since the departure of Schulla from ETH, the model is now known simply as WaSIM. In the original model, runoff generation was based on the TOPMODEL approach (Beven and Kirkby, 1979). Recent model version updates have incorporated a layered soil model based on the Richards' equation, a multi-layer groundwater model, layered vegetation, an irrigation module, dynamic plant phenology, a glacier model, interface for on-line-coupling with other models, a lake module, and surface discharge routing (Table 4-7). High meteorological input requirements are needed for the model to run in a fully physically-based configuration. Two freeware versions are available, either

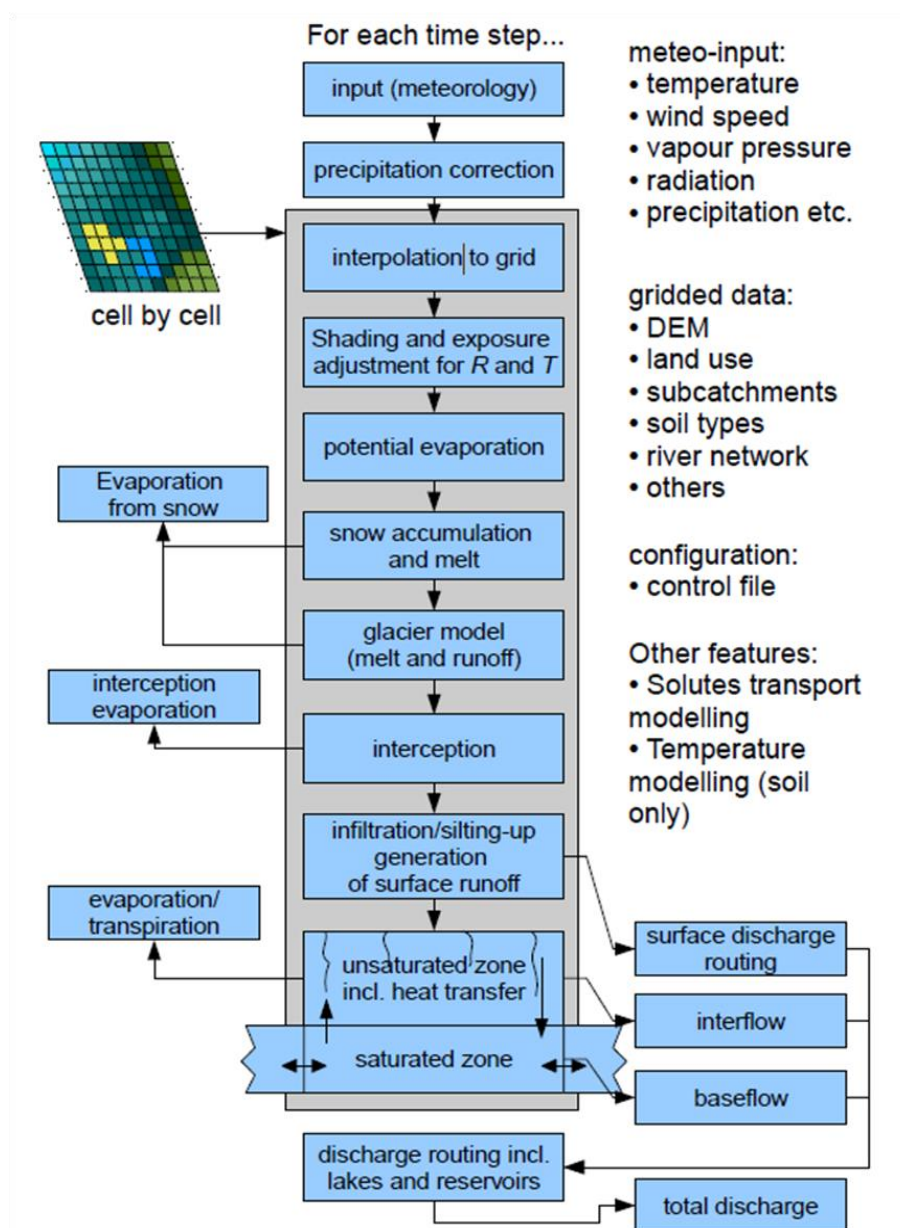


Figure 4-4. WaSIM-ETH model structure (Schulla and Jasper, 2012)

Table 4-7. WaSIM model capability against rating criteria

Criterion	Model capability
Physical representation	Interception: Rainfall fraction and maximum storage threshold Evapotranspiration: After Penman-Monteith (Monteith, 1965), Wendling, Hamon or Haude dependent on time step. Snowmelt: Temperature index and energy balance approach Overland flow: 2D diffusive wave Unsaturated zone flow: TOPMODEL or 3D variably saturated Richards equation Groundwater flow: 3D variably saturated Richards equation, reduces to Darcy flow when fully saturated Channel flow: 1D kinematic wave
Distribution	Fully distributed, finite difference.
Integration	Surface and subsurface domains solved separately yet iteratively at the same time step.
Ease of use	Modular system. GUI system exists for Richards-version. Otherwise runs from command line and batch files. Import of spatial data with GIS (ArcGIS)
Applicability	Krause and Bronstert (2004)
Calibration	Manual or external programme
Temporal representation	Runs in constant time steps, although subroutines may run at different sub time steps. Automatic time step algorithm.
Scale	Grid size from centimetres to several kilometres. Applications in small to large watersheds.
Training and cost	Source code available. Manual and tutorial available. Model support from developers.

using the TOPMODEL approach for the runoff generation and soil simulation, or using the Richards approach for the unsaturated zone and runoff generation. These are both available for single processor and multiprocessor systems.

4.2.2 Performance of hydrological modelling systems against requisites

The outputs from model capability ratings against selection criteria are presented in Table 4-8. Initial general trends may be seen, with low performance around calibration, and high performance in the field of training and cost. MIKE SHE evidently attracts the greater number of High ratings, whilst Medium ratings dominate SHETRAN and WaSIM capabilities. HGS and MODHMS display a more even spread of ratings through the selection criteria.

Applying a simple scoring method (Low=0, Medium=1, High=2) to the total ratings allows the models to be ranked according to overall suitability (Table 4-9). MIKE SHE displays

Table 4-8. Model ratings by selection criteria

Criterion	HGS	MIKE SHE	MODHMS	SHETRAN	WaSIM
Physical representation	L	H	M	M	H
Distribution	H	M	M	M	M
Integration	H	M	H	M	M
Ease of use	M	H	H	M	M
Applicability	M	H	L	L	M
Calibration	L	H	M	L	L
Temporal representation	M	H	L	M	M
Scale	H	H	H	M	M
Training	L	H	M	H	H

Table 4-9. Model ratings and scores

Model	Ratings			Score
	L	M	H	
HGS	3	3	3	9
MIKE SHE	0	2	7	16
MODHMS	2	4	3	10
SHETRAN	2	6	1	8
WaSIM	1	6	2	10

a noticeable higher score than the other models, which are otherwise closely ranged with SHETRAN scoring least.

Grouping the scores within the evaluative dimensions of functionality, adaptability and accessibility allows the models to be positioned on a matrix (Figure 4-5). Axes originate at 2 for illustrative purposes.

Aside from SHETRAN, the models rate equally in terms of functionality. Differences appear when considering adaptability and accessibility. The close grouping of HGS, MODHMS, SHETRAN and WaSIM compared to MIKE SHE is conspicuous.

4.2.3 Selection of the MIKE SHE hydrological modelling system

As a result of the above review of potential models, the MIKE SHE Release 2009 modelling system was selected for use in this study. A noticeably higher rating reflects an overall compatibility of the modelling system within the range and limits set by the

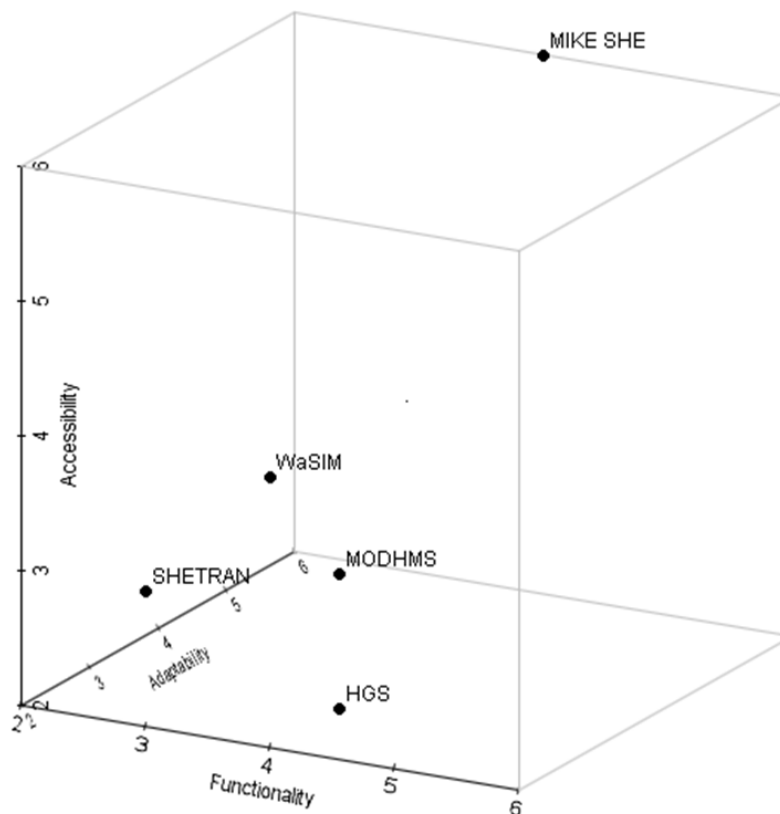


Figure 4-5. Model position on matrix of functionality, adaptability and accessibility

selection criteria, and justify its choice for this study. The other models are relatively closely ranked, although each has strengths in differing areas. For the research needs, the versatility and adaptability of MIKE SHE are compelling.

An intuitive GUI, ability to incorporate GIS data and availability of support defines MIKE SHE against models such as HGS and WaSIM. A lack of proven applicability to wetland hydrology and poor temporal representation are a hindrance for MODHMS, despite a foundation upon the widely acknowledged MODFLOW code. The weakest model reviewed for the purposes of this study, SHETRAN, is more suited to sediment and solute transport, although fully integrated hydrologically.

HGS and MODHMS are able to provide sophisticated couplings of the surface and subsurface flow domains, with representative equations solved simultaneously for each time step. However, with such an increase in complexity comes an associative demand on computational power, raising model runtimes. It is considered that the coupling in MIKE SHE/MIKE 11 is acceptable with all criteria taken into account. The models perform equally in terms of overall functionality.

The ability to increase the complexity of process representation in MIKE SHE iteratively as more data and understanding become available is highly appealing. The modular framework allows each process to be solved at a distinct and appropriate time step and spatial scale, reducing computational demand. This flexibility, along with a proven applicability for simulating wetland hydrological systems (Al-Khudhairy et al., 1999; Duranel, 2015; Refsgaard et al., 1998; Staes et al., 2009; Thompson et al., 2009; Thompson et al., 2004), underpins its use in the current study.

4.2.4 Process representation in MIKE SHE

The modular process-based approach, introduced above, used within MIKE SHE enables multiple representations for each hydrological process (Figure 4-6). This feature is relevant when considering the limitations of physics-based model code, where partial differential equations of mass flow and momentum transfer are solved. These include large data requirements, lengthy computation time, over-parameterisation and transferability of equations to different scales (Graham and Butts, 2005). Simplified process representation is advantageous, for example, when modelling large catchments where the water balance is dominated by few processes (Andersen et al., 2001; Henriksen et al., 2003; Vázquez et al., 2002). However, when simulating groundwater and surface water flow a more complex process representation is required (Madsen and Kristensen, 2002; Refsgaard et al., 1998; Sonnenborg et al., 2003). The hydrological processes relevant to this study and the representative methods available within MIKE SHE are outlined below.

Precipitation

Precipitation is the main direct input into MIKE SHE. Rate of precipitation may be specified as a constant value or time series with either uniform, station-based using, for example, Thiessen polygons to distribute the extent of influence of each station, or full gridded distribution. If snowmelt is included, air temperature is supplied in the same format. Snowmelt is described with a modified degree-day method after the HBV model (Bergström, 1975). Rate of melting is related to air temperature, extended to solar radiation and rain-on-snow.

Evapotranspiration

In MIKE SHE, actual evapotranspiration is calculated from interception, soil, ponded water, the canopy surface, root zone and groundwater. Vegetation properties and meteorological data on potential evapotranspiration are required inputs. A proportion of

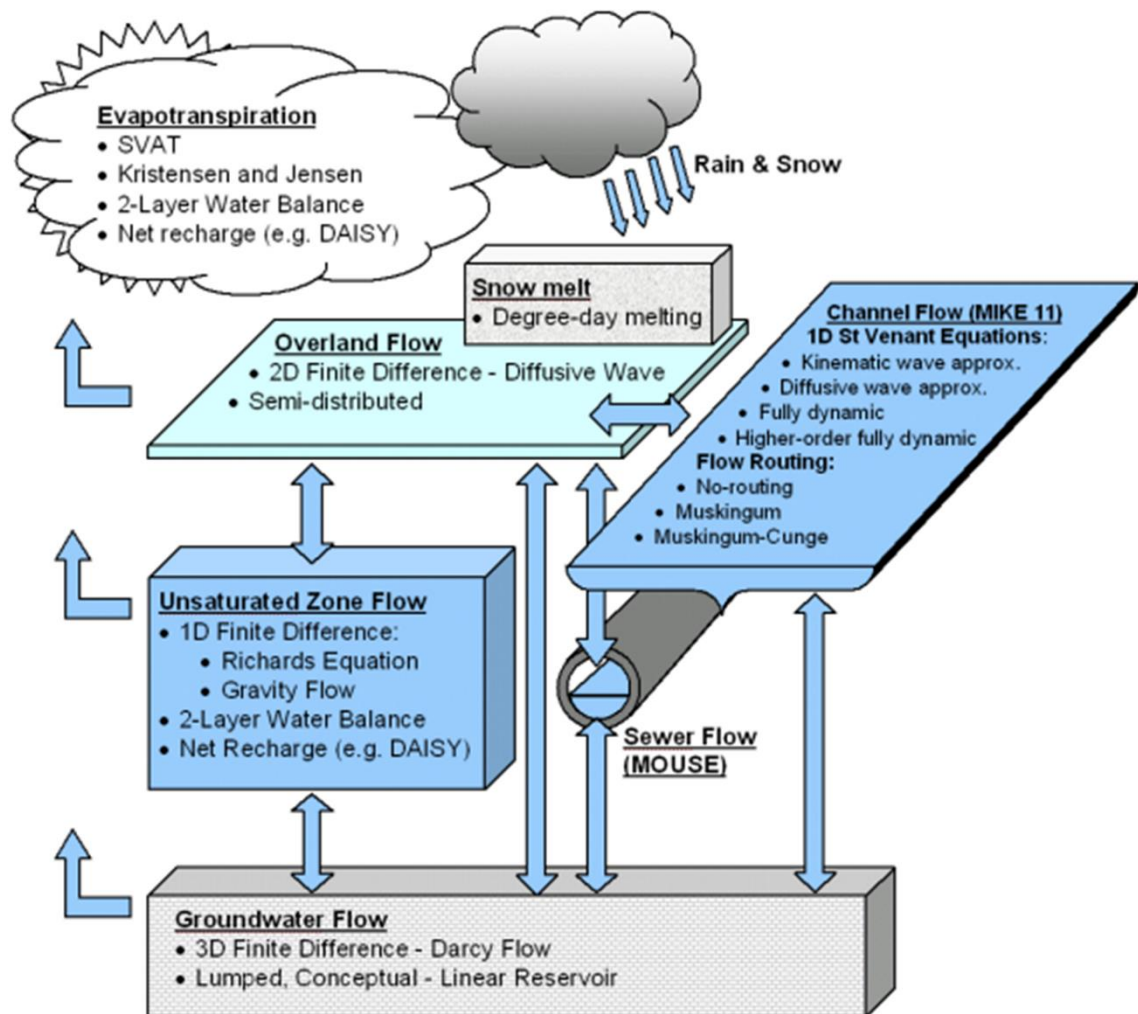


Figure 4-6. Schematic of process representation in MIKE SHE, showing the available numerical solvers and exchange pathways (Graham and Butts, 2005)

the rainfall is intercepted and evaporated from the vegetation canopy, dependent on surface characteristics of vegetation and leaf area index (LAI). Water reaching the soil surface either produces runoff or infiltrates to the unsaturated zone. The infiltrating water is part evaporated from the upper root zone or transpired by plant roots, with the remainder going to groundwater recharge. Actual evapotranspiration are calculated by one of four methods:

- Soil Vegetation Atmosphere Transfer (SVAT)

A two-layer land surface model of the soil and canopy is linked together by a network of resistances (Shuttleworth and Gurney, 1990; Shuttleworth and Wallace, 1985). A single canopy layer is located above the soil layer. Water may only leave the soil layer through the canopy layer, with fluxes of evaporation and heat driven by differences in humidity and temperature.

- Kristensen and Jensen

Using empirical data, Kristensen and Jensen (1975) derived a method for indirect estimation of actual evapotranspiration as a function of measurable values of potential evapotranspiration. Actual evapotranspiration is calculated on the basis of potential evapotranspiration and some reduction functions derived from specified root depth, LAI and calculated soil moisture. The potential evapotranspiration input term may be derived from pan measurements, or by use of alternative analytical methods (Thompson et al., 2014).

- Two-layer water balance

This simplified water balance method uses a soil layer approach to calculating actual evapotranspiration based upon the soil moisture profile (Yan and Smith, 1994). The unsaturated zone is divided into a root zone, from which evapotranspiration can occur and a zone below the root zone, where evapotranspiration does not occur.

- External software

Mainly for agricultural purposes, external programmes may be used with MIKE SHE, such as DAISY, a soil-plant-atmosphere model (Abrahamsen and Hansen, 2000; Hansen et al., 1990). Such models are used to simulate changes in crop yield and when used with MIKE SHE, replace the evapotranspiration and unsaturated zone processes.

Overland flow

Overland flow on the ground surface results from ponded water routed downhill. This may occur, for example, from precipitation that has not infiltrated into the unsaturated zone, groundwater seepage, or overbank spillage from streams. Flow direction and rate is determined by topography and resistance, with losses due to evaporation, infiltration and overbank to channel flow. Overland flow in MIKE SHE may be calculated using either a finite-difference, diffusive wave approximation of the Saint Venant equations, or a semi-distributed, slope-zone approach:

- 2D Finite Difference

Realistic representation of free surface flow requires use of the governing equations of fluid dynamics. In general form, fluid motion is described by the time-dependent 3D Navier-Stokes equations, and comprise coupled differential

equations of conservation of momentum and mass. However, due to the high computational requirements and length of calculation time needed to solve the Navier-Stokes equations, a common method for overland flow, and that implemented in MIKE SHE, is to use the St Venant equations in 2D form. These are obtained by depth averaging the Navier-Stokes equations, providing a sheet flow approximation. Surface friction slopes are expressed by the Manning equation.

- Semi-distributed

The semi-distributed approach for overland flow in MIKE SHE is based on an empirical relationship between flow depth and surface detention, with turbulent flow described by the Manning equation. The method is similar to the Stanford Watershed Model (Crawford and Linsley, 1966) and uses a simplified representation of overland flow within topographical zones. Flow between zones, from the catchment headlands to streams, is conceptualised as a cascade of overland flow areas.

Unsaturated zone flow

In the unsaturated zone it is reasonable to assume that flow occurs in the vertical plane only, due to the principal role of gravity in infiltration. Although the assumption loses validity for steep slopes, or small-scale models with lateral flow, to reduce excessive computation time MIKE SHE only calculates unsaturated flow vertically. Four solution methods are available within MIKE SHE for calculating unsaturated flow:

- Richards equation

The 1D form of the Richards equation is the most accurate, yet computationally demanding, method for dynamic unsaturated flow. Though, to reduce computation demands, calculation nodes can be distributed. The pressure head and hydraulic conductivity are required as functions of saturation. Hence, descriptions of the relationships for the soil moisture retention curve and effective conductivity are needed for input. The Richards equation is calculated with a finite difference solution (Refsgaard and Storm, 1995). Evapotranspiration in the upper soil layer and root zone, and an iterative coupling to the saturated zone provide the boundaries.

- Gravity flow

Gravity flow is a simplified version of the Richards equation with the pressure head term omitted. The function for downward flow becomes solely gravity, with capillary action ignored. This is of use when the main focus is on groundwater recharge based on precipitation and evapotranspiration rather than flow dynamics in the unsaturated zone, and for coarse soils.

- Two-layer water balance

As with the two-layer water balance method for evapotranspiration, the unsaturated zone is divided into a root zone and a zone below the root zone. Infiltration is routed to the saturated zone when unsaturated storage becomes zero. This method is suitable when the water table is shallow, groundwater recharge is dominated by evapotranspiration in the root zone, and the time for water moving through the unsaturated zone is of no interest (Graham and Butts, 2005).

- Net recharge

Recharge may be input directly to the saturated zone, typically when external programmes are used for modelling interactions between soil, plants and atmosphere.

Groundwater flow

Representation of groundwater flow in MIKE SHE is by either a 3D finite difference method, or a simpler linear reservoir approach. These handle different, often scale dependent, requirements for detail. Models below catchment scale for impact assessments often call for detailed treatment of groundwater/surface water interaction. On the other hand, water resource applications at catchment scale and above generally only need information on water balances and trends:

- Darcy flow

For saturated groundwater flow in three dimensions, the 3D Darcy equation may be used to describe variations in hydraulic head. This is solved iteratively by a finite difference method, for which two solvers are available in MIKE SHE: the successive over-relaxation (SOR) technique and a preconditioned conjugate gradient (PCG) technique identical to that used within MODFLOW (Hill, 1990; McDonald and Harbaugh, 1988).

- Linear reservoir

A conceptual approach, the linear reservoir method provides a compromise between data limitations and model simplicity. The groundwater catchment, or model domain, is divided into sub-catchments. These are divided into a series of shallow interflow reservoirs, with one or two deeper baseflow reservoirs. Groundwater is routed through the linear reservoirs as interflow and baseflow.

Channel flow

In MIKE SHE it is assumed that streams are one-dimensional, due to the detailed bathymetric data and computational burden needed to calculate two-dimensional surface flow. This is deemed reasonable for most cases and results in uniform discharge and stage across the channel. 1D channel flow is calculated in MIKE SHE by the hydraulic modelling system MIKE 11, which models unsteady flow and water levels in channels using either a finite difference approach or a simple flow routing method:

- 1D St Venant equations

For applications where a detailed knowledge of flow dynamics is needed, the 1D form of the St Venant equations are used to calculate channel flow. In MIKE 11 these are solved either in full non-linear form or by using a diffusive wave, kinematic wave, or quasi steady state approximations. An ability to calculate subcritical and supercritical flow by these methods enables simulation of flow through a wide range of structures, such as weirs, bridges and regulating or control structures.

- Flow routing

An alternative method to represent channel flow is lumped flow routing, whereby the flow hydrograph may be determined at a point on the channel from a known upstream hydrograph. Governed by a continuity equation and flow-storage relationship, flow is calculated as a water pulse through the system by means of a transfer function. In MIKE 11 flow routing procedures range from instantaneous flow routing to the simplistic Muskingum approximations and Muskingum-Cunge approaches, where Muskingum routing parameters are related to characteristics including channel length, flood wave celerity, unit width discharge and channel bed slope (Birkhead and James, 2002).

- MIKE SHE / MIKE 11 coupling

The MIKE 11 river network is composed of digitised points and computational nodes, which are interpolated to the MIKE SHE grid (Graham and Butts, 2005; Thompson et al., 2004) (Figure 4-7). Exchange by overbank flow or between the saturated zone and channels, which can be bidirectional depending on levels, occurs across the edges between grid cells. Hence, spatial accuracy is directly related to the grid resolution. MIKE SHE limits exchange to a sub-set of the MIKE 11 river network that intersects the MIKE SHE grid. River link locations in MIKE SHE are located automatically dependent on the co-ordinates that define the MIKE 11 network. MIKE 11 also includes the ability for alternative bed materials to be specified. Exchange between the saturated zone and the river can depend on the hydraulic conductivity of the aquifer material only, the conductivity of the river bed material only, or the conductivity of both the river bed and the aquifer material (DHI, 2009b).

4.3 Model development

The model domain was provided by the River Lambourn Observatory formal boundary with a total area of 10 ha (Figure 4-8). This coincides with the perimeters of the wetland areas at the extents of the valley bottom. A grid size of 1 × 1 m was selected from a series of model runs which showed little change in simulated groundwater heads for grid

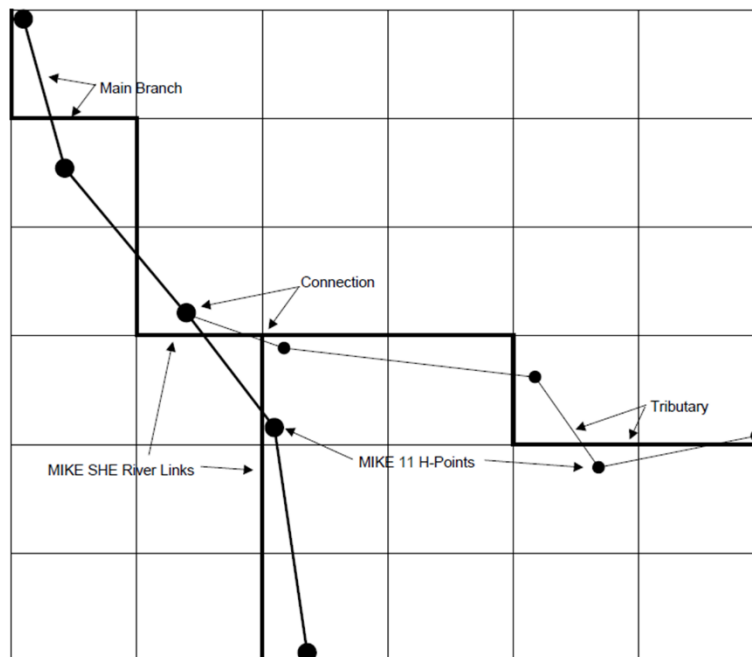


Figure 4-7. MIKE 11 branches and calculation points in a MIKE SHE grid with river links (DHI, 2009a)

sizes between 0.5×0.5 m and 10×10 m (after Vázquez et al. (2002)). The chosen resolution, which produces 101,689 grid cells within the model domain, provides a good balance between the representation of physical characteristics of the site, such as topography, and computation time. Model time step was adjusted automatically within MIKE SHE dependent on precipitation and infiltration rates (DHI, 2009b), with the maximum time step set at 24 hours. The computational time for each model run was approximately 2 hours.

Detailed topographic data were provided by the combined ground survey and LiDAR (Section 3.3), resampled to the 1×1 m MIKE SHE grid (Figure 4-9). A single long grass vegetation type was used to represent land cover across the model in line with the dominance of tall-herb fen at the site (House et al., 2015). Temporal variations in leaf area index and root zone depth, required for the interception and evapotranspiration modules, were taken from Breuer et al. (2003) and an existing DHI (2009b) vegetation

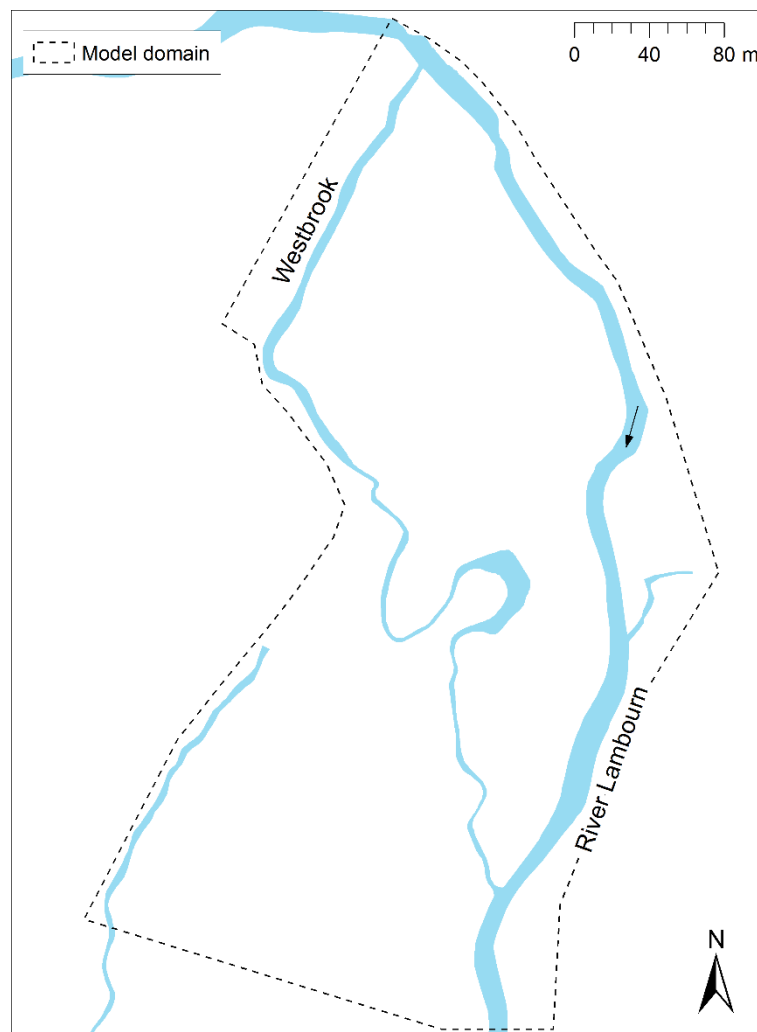


Figure 4-8. MIKE SHE model domain boundary

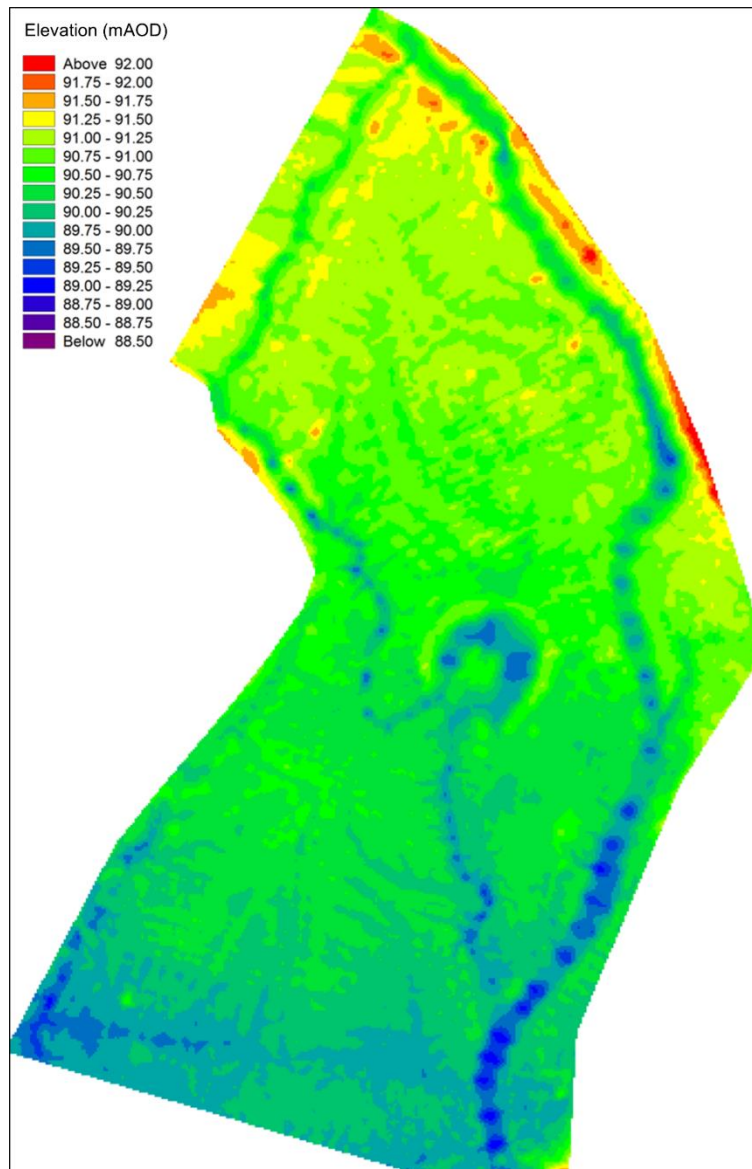


Figure 4-9. MIKE SHE 1 x 1 m topographic grid of the Observatory

properties file. The overland flow component was uniformly distributed, with the Manning's n roughness coefficient employed as a calibration term. Numerical errors in solving overland flow were reduced through specification of the Explicit Numerical Solution method, which calculates flow based on individual cell heads.

Unsaturated flow was calculated in a reduced number of cells, subsetting automatically and dynamically within MIKE SHE based on depth to groundwater. The 1D form of the Richards equation used within the unsaturated zone module employed a spatially uniform soil profile comprising peat to a depth of 1 m. Values for the Van Genuchten (1980) expression of the soil moisture retention curve were obtained from Letts et al. (2000). Saturated hydraulic conductivity and effective saturation were parameterised through calibration.

The saturated zone was characterised as a four layer geological model with peat overlying gravels over a discontinuous layer of putty chalk, and Chalk bedrock beneath. The 3D finite difference Darcy flow method was employed to calculate subsurface flow. Depths to the gravel-peat interface were taken from a manual probing survey of 2815 locations (Section 2.5.2) in conjunction with the topographic survey (Chambers et al., 2014). The gravel-chalk interface was derived from a 3.1 ha 3D ERT survey of the meadows using a resistivity isosurface extended by trilinear interpolation from intrusive boreholes where the interface could be identified in core retrievals (Chambers et al., 2014) (Section 2.5.3). This was extended to the edges of the model domain by bilinear interpolation within MIKE SHE. The horizontal extent of the putty chalk was specified to follow the extraction from the resistivity model (Section 3.10), with an additional absence at the northern tip of the site to follow the temperature anomaly and area of persistently high groundwater head elevations (see Section 3.11.1) (Figure 4-10).

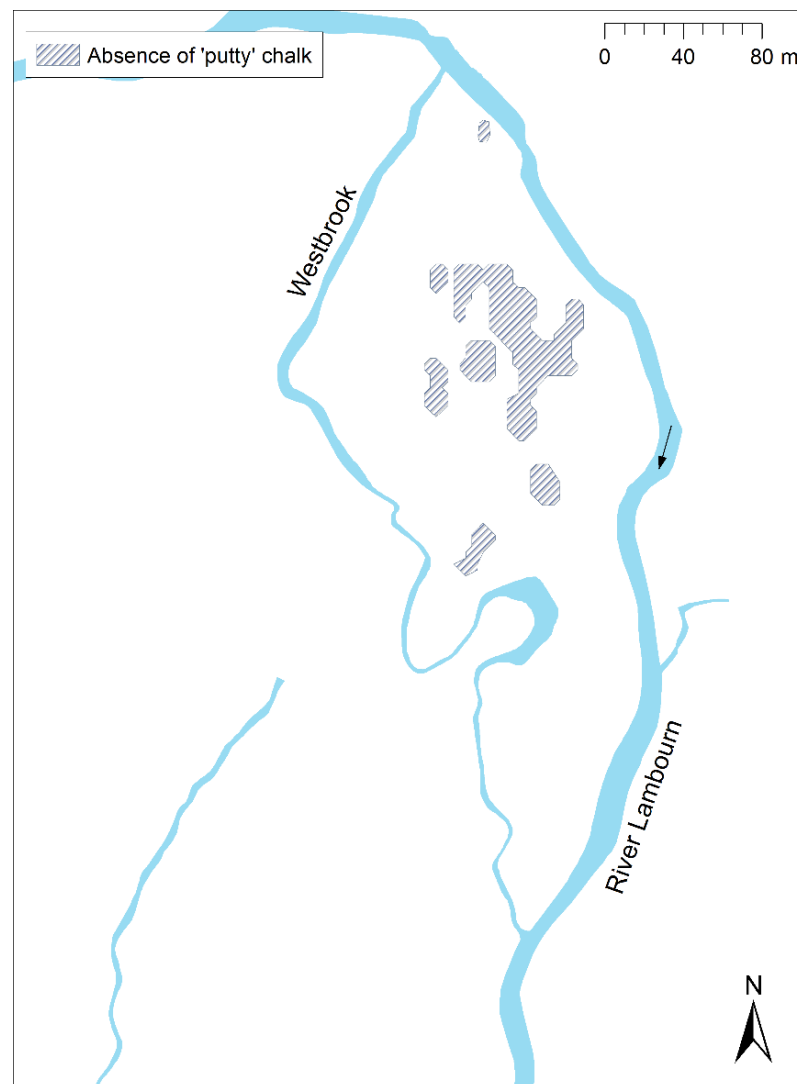


Figure 4-10. Horizontal extent of MIKE SHE putty chalk geological layer

Vertical and horizontal hydraulic conductivities of the peat and gravels were employed as calibration terms. The model was also sensitive to vertical hydraulic conductivity in the Chalk, which formed a manual calibration parameter in the initial model build as it predominantly affected bias. This was finalised at 0.00438 ms^{-1} , an order of magnitude above the horizontal hydraulic conductivity taken from the literature as $4.4 \times 10^{-4} \text{ ms}^{-1}$ (Younger, 1989). Horizontal extents of the putty chalk were represented as a discontinuous 1 m layer at the top of the Chalk bedrock. Hydraulic conductivity was specified as $1 \times 10^{-6} \text{ ms}^{-1}$ in line with the literature (Younger, 1989), although model performance did not vary below this value. Gaps in the putty chalk were allocated the same hydraulic conductivity as the Chalk. For the Chalk aquifer, head boundaries were based on observations from piezometer 3C. These were adjusted to differences in elevation along the model boundary by linear interpolation. The assumption that the boundary condition in the chalk varies with terrain elevation was considered reasonable given the site's riparian position and relatively flat topography. Gravel boundaries were set to a constant flux gradient of 0.003 m in the north and south, following the topographic gradient, with the remaining boundaries defined as zero flow. Zero flow boundaries were assigned around the peat and putty chalk layers where lateral flow was assumed to be minimal due to low hydraulic conductivities.

The river network was digitised in MIKE 11 from Ordnance Survey MasterMap 1:1250 raster data (Figure 4-11). The fully dynamic 1D St Venant equations were used to describe channel flow with all MIKE 11 branches specified as being coupled to MIKE SHE. Channel cross-section profiles applied to the network were based on the dGPS surveys conducted at 44 locations along the Westbrook and 42 along the River Lambourn (Section 3.5.1) (Figure 4-11). Bank elevations were taken from the $1 \times 1 \text{ m}$ MIKE SHE topographic grid with points across the cross-section specified as depths relative to the banks (Thompson et al., 2004). The channel bed was specified to be in full contact with the saturated zone, so that exchange between the river and aquifer was controlled by the hydraulic conductivity of the aquifer rather than river bed material. This was deemed appropriate due to the nature of the channel substrate and high base flow index.

Inflows for the upstream channel boundary, which were specified as a mean 15 minute discharge, were taken from the derived relationship between monthly measurements of discharge at L1 using an electromagnetic flow meter and the corresponding flow at the downstream Shaw gauging station (Section 3.5.2). The downstream boundary was set to follow monthly stage observations at L7 specified on a 15 minute basis by linear interpolation.

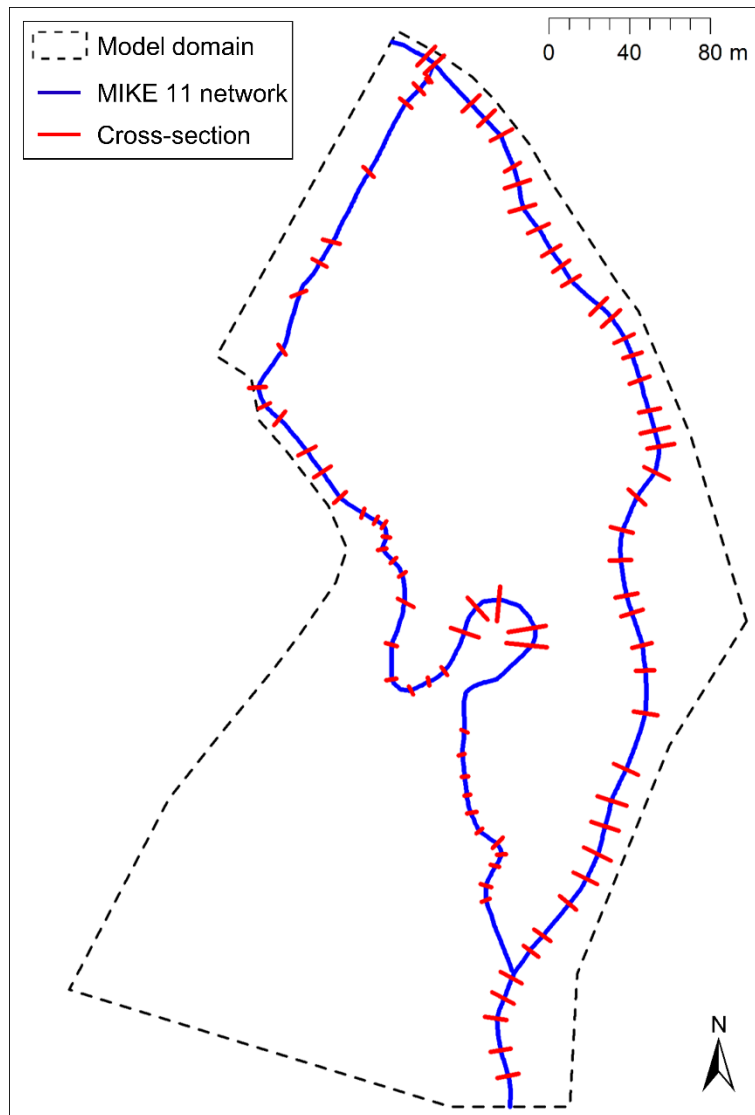


Figure 4-2. MIKE SHE model domain showing MIKE 11 river network and channel cross-sections

MIKE 11 does not contain a method to explicitly represent volumetric and temporal changes in instream vegetation. To account for macrophyte growth and its removal by cutting within the Lambourn the hydraulic resistance, fundamental to the depth-discharge relationship, was adjusted as a proxy. Hydraulic resistance was expressed as a 15 minute time series of Manning's n coefficients (Figure 4-12). Values were derived from measurements of cross-section geometry and stage at L1, energy slope between stage boards at L1 and L2, and the derived 15 minute discharge at L1. The time series was applied to the entire reach as a multiplication factor to a fixed channel roughness, with the assumption that variations in macrophyte growth were uniform along the reach. Manning's n values specified in this way fluctuated between 0.045 and 0.353 in response to the growing season. Increases in discharge from storm events generally correspond

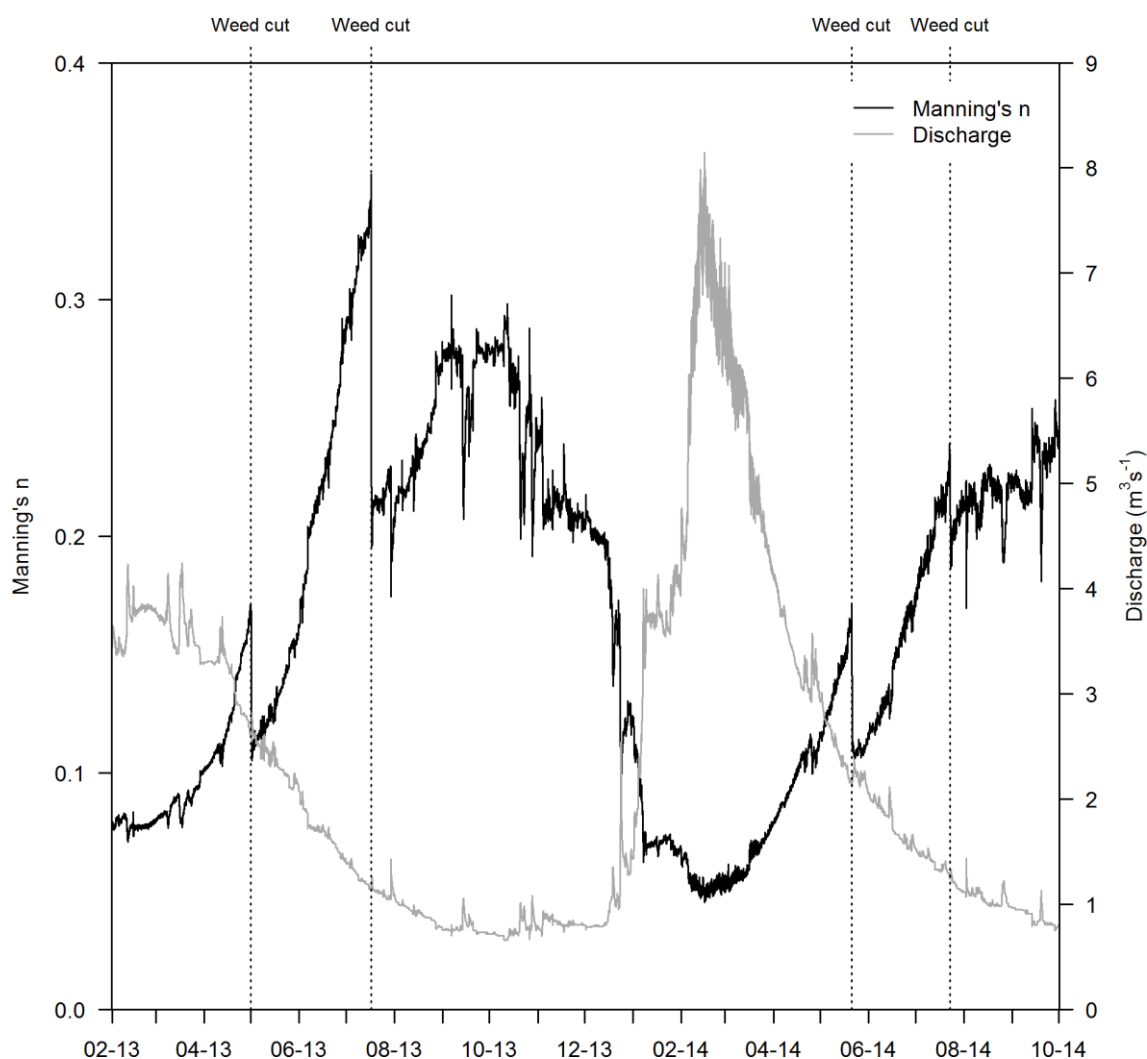


Figure 4-12. Calculated Manning's n roughness coefficient and discharge inputs for the MIKE 11 hydraulic model

to decreases in Manning's n . This is assumed to be due to the flattening or removal of vegetation by the high flows. However, weed cuts within the channel undertaken on 1/5/2013, 16/7/2013, 21/5/2014 and 23/7/2014 caused rapid drops in Manning's n with no equivalent change in discharge.

Meteorological data were supplied by the automatic weather station installed in the south meadow. This provided 15 minute precipitation (Section 3.4.1) and potential evapotranspiration calculated using the Penman-Monteith formula (Monteith, 1965) (Section 3.4.2). MIKE SHE calculated actual evapotranspiration from these specified potential rates and computed soil moisture in the root zone using the Kristensen and Jensen (1975) method (see earlier Section 4.2.4).

4.4 Calibration and validation

The periods 1/2/2013–1/12/2013 and 1/12/2013–1/10/2014 were used for split sample calibration and validation, respectively. This was based on comparisons between simulated and observed head elevations in the peat and gravel piezometers installed at the site. Calibration and validation of channel stage was based on comparisons between MIKE 11 simulated stage and observations from stage boards. In accordance with the literature (Refsgaard and Storm, 1995), the number of calibration parameters was minimized. As mentioned above, parameters adjusted during calibration included effective saturation and infiltration rate in the unsaturated zone, the vertical and horizontal hydraulic conductivity of the peat and gravel in the saturated zone, and Manning's n roughness coefficient for overland flow.

An automatic multiple objective calibration was performed based on the shuffled complex evolution method (Duan et al., 1992; Madsen, 2000; Madsen, 2003). Model performance statistics comprised the root mean square error (RMSE) for goodness of fit and the absolute value of the average error for bias. The calibration problem is solved by defining a single objective function that aggregates the different objective functions into a single statistic (Madsen, 2003). The auto-calibration routine was run until convergence criteria were met, in this case when the minimum relative change in the aggregated objective function was less than or equal to 0.01. This required 88 simulations with a computation time of approximately 1 week.

Manual adjustment of calibration parameters to further improve model performance was assessed using the Pearson correlation coefficient (R), the Nash-Sutcliffe coefficient (R^2) (Nash and Sutcliffe, 1970) and the root mean square error (RMSE) of the deviation between observed and simulated groundwater and channel water levels. A scheme adapted from Henriksen et al. (2008), who used a point based system for different classes of performance indicators, was used to classify model performance based on the values of these statistics.

4.4.1 Simulated and observed values

Final values for the seven calibration parameters indicate values of horizontal and vertical hydraulic conductivity of the peat are consistent with the scale-dependency of peat hydraulic conductivity reported by Bromley et al. (2004) (Table 4-10). Higher values are obtained with increasing volumes due to preferential flow routes provided by features such as root holes and abandoned infilled ditches. Gravel hydraulic conductivity values are supported by similar measurements from superficial deposits throughout the Thames

Table 4-10. Calibrated parameter values

Parameter	Calibrated value
Unsaturated zone effective saturation	0.93
Unsaturated zone infiltration rate (ms ⁻¹)	2.30 × 10 ⁻⁵
Peat horizontal hydraulic conductivity (ms ⁻¹)	1.98 × 10 ⁻⁵
Peat vertical hydraulic conductivity (ms ⁻¹)	9.53 × 10 ⁻⁶
Gravel horizontal hydraulic conductivity (ms ⁻¹)	2.93 × 10 ⁻⁴
Gravel vertical hydraulic conductivity (ms ⁻¹)	6.98 × 10 ⁻⁴
Manning's n coefficient for overland flow (sm ^{-1/3})	0.03

Basin (Bricker and Bloomfield, 2014). Model performance statistics for the calibration period for all 20 piezometers and the ten stage boards show that, according to the classification scheme, model performance is generally “very good” to “excellent” (Table 4-11). Mean values for RMSE, R and R² are 0.063 m (“very good”), 0.92 (“excellent”) and 0.75 (“very good”) respectively. Model results are notably better for the gravel groundwater heads compared to those in the peat. Out of 30 values for the gravel (ten piezometers × three statistics), 23 are classified as “excellent” with the remainder classed as “very good”. In contrast, 15 of the 30 values for the peat piezometers are classed as “excellent” and ten as “very good”. Four of the remaining five values are classed as “fair” whilst the R² value for 2P (0.35) is “poor” (although the RMSE and R values are “very good” and “excellent”, respectively). Model performance for channel stage is predominantly “very good” (16 out of 30 values) followed by “excellent” (10 values) although three R² values and one RMSE value are classified as only “fair”.

Performance for the validation period is in general very similar to the calibration period (Table 4-12). The mean values for RMSE, R and R² are 0.063 m (“very good”), 0.94 (“excellent”) and 0.77 (“very good”), respectively. For the gravel piezometers 16 of the statistics are classified as “excellent” with the remainder being “very good”. Slightly more of the statistics have a higher value for the calibration period than the validation (16:10 with four unchanged). For the peat piezometers, an equal number (13) of the statistics are classed as “excellent” or “very good” with two each being classified as “fair” or “poor”. Performance as indicated by these statistics is improved for the validation period for 12 and reduced for 17 but in general changes are small in magnitude (one value remaining the same). The previous classification of R² as “poor” for P2 is replaced by a “very good” whilst the same statistics for 5P and 8P, which for the calibration period were classed as “very good”, are now “poor”. A marked improvement in the model’s ability to simulate channel stage for the validation period is evident with 19 of the 30 statistics having higher

Table 4-11. Model performance statistics for the calibration period (1/2/2013 – 1/12/2013).

Model performance indicators are adapted from Henriksen et al. (2008)

Observation sites	RMSE (m)	R	R2
1G	0.020 *****	0.99 *****	0.98 *****
1P	0.055 *****	0.98 *****	0.78 *****
2G	0.026 *****	0.99 *****	0.97 *****
2P	0.082 *****	0.87 *****	0.35 **
3G	0.067 *****	0.98 *****	0.78 *****
3P	0.047 *****	0.96 *****	0.86 *****
4G	0.047 *****	0.98 *****	0.90 *****
4P	0.122 ***	0.92 *****	0.57 ***
5G	0.059 *****	0.91 *****	0.82 *****
5P	0.069 *****	0.86 *****	0.74 *****
6G	0.076 *****	0.95 *****	0.74 *****
6P	0.100 *****	0.92 *****	0.61 ***
7P	0.095 *****	0.81 *****	0.63 ***
8G	0.034 *****	0.99 *****	0.91 *****
8P	0.029 *****	0.94 *****	0.80 *****
9G	0.021 *****	0.99 *****	0.97 *****
9P	0.034 *****	0.99 *****	0.88 *****
10P	0.035 *****	0.98 *****	0.74 *****
11G	0.036 *****	0.99 *****	0.89 *****
12G	0.041 *****	0.97 *****	0.84 *****
L1	0.078 *****	0.87 *****	0.71 *****
L2	0.035 *****	0.97 *****	0.91 *****
L3	0.086 *****	0.84 *****	0.69 *****
L4	0.082 *****	0.84 *****	0.68 *****
L5	0.072 *****	0.86 *****	0.72 *****
L6	0.075 *****	0.88 *****	0.64 ***
L7	0.064 *****	0.88 *****	0.73 *****
W1	0.078 *****	0.85 *****	0.67 *****
W2	0.110 ***	0.87 *****	0.63 ***
W3	0.086 *****	0.93 *****	0.63 ***
Performance indicators			
Excellent *****	<0.05	>0.85	>0.85
Very good ****	0.10-0.05	0.65-0.85	0.65-0.85
Fair ***	0.15-0.10	0.50-0.65	0.50-0.65
Poor **	0.20-0.15	0.20-0.50	0.20-0.50
Very poor *	>0.20	<0.20	<0.20

values for latter period (ten lower, one unchanged). Performance is predominantly classified as “excellent” (22 statistics). With the exception of RMSE for W2 (“fair”), the others are classified as “very good”.

Table 4-12. Model performance statistics for the validation period (1/12/2013 – 1/10/2014).

Model performance indicators are adapted from Henriksen et al. (2008)

Observation sites	RMSE (m)	R	R2
1G	0.057 ****	0.98 *****	0.86 *****
1P	0.037 *****	0.97 *****	0.92 *****
2G	0.075 ****	0.97 *****	0.75 ****
2P	0.059 ****	0.96 *****	0.67 ****
3G	0.080 ****	0.98 *****	0.66 ****
3P	0.070 ****	0.93 *****	0.57 ***
4G	0.030 *****	0.98 *****	0.96 *****
4P	0.106 ***	0.92 *****	0.70 ****
5G	0.094 ****	0.87 *****	0.66 ****
5P	0.099 ****	0.76 ****	0.46 **
6G	0.072 ****	0.92 *****	0.76 ****
6P	0.082 ****	0.85 *****	0.70 ****
7P	0.093 ****	0.85 *****	0.68 ****
8G	0.079 ****	0.98 *****	0.82 ****
8P	0.058 ****	0.89 *****	0.31 **
9G	0.056 ****	0.99 *****	0.85 *****
9P	0.059 ****	0.96 *****	0.72 ****
10P	0.034 *****	0.96 *****	0.87 *****
11G	0.027 *****	0.99 *****	0.97 *****
12G	0.088 ****	0.89 *****	0.72 ****
L1	0.052 ****	0.97 *****	0.85 *****
L2	0.027 *****	0.98 *****	0.92 *****
L3	0.048 *****	0.97 *****	0.87 *****
L4	0.062 ****	0.97 *****	0.79 ****
L5	0.049 *****	0.94 *****	0.86 *****
L6	0.061 ****	0.98 *****	0.85 *****
L7	0.032 *****	0.98 *****	0.96 *****
W1	0.047 *****	0.97 *****	0.86 *****
W2	0.101 ***	0.87 *****	0.73 ****
W3	0.069 ****	0.98 *****	0.81 ****
Performance indicators			
Excellent *****	<0.05	>0.85	>0.85
Very good ****	0.10-0.05	0.65-0.85	0.65-0.85
Fair ***	0.15-0.10	0.50-0.65	0.50-0.65
Poor **	0.20-0.15	0.20-0.50	0.20-0.50
Very poor *	>0.20	<0.20	<0.20

The generally “excellent” or “very good” performance of the model in terms of reproducing the observed gravel groundwater head elevations are shown throughout both the calibration and validation periods (Figures 4-13 and 4-14). The simulated heads clearly display the seasonal rise and fall observed throughout the site as well as the

impacts of individual rain events. In addition, the effects of the weed cuts in the form of the subsequent rapid declines in groundwater head are clearly simulated by the model suggesting good representation of the exchange between the river and the underlying gravels. Some over prediction towards the end of the validation period is noticeable at 1G, 2G and 5G (Figure 4-13). Groundwater heads at some locations with upwelling (8G, 9G and 12G) are under predicted during the period of high head elevation at the beginning of 2014 although at 11G this over estimate of head elevation is not apparent (Figure 4-14). Weaker performance is apparent at 3G and 6G, with under prediction notable during periods of high head.

As noted previously, model performance for the peat groundwater head elevations is inferior to the gravels (Figures 4-15 and 4-16). The impacts of many of the individual rain events are, however, simulated as are the rapid declines in level associated with the weed cuts in the River Lambourn. Model performance tends to be better at low head elevations. Nonetheless, observed peat groundwater head at locations P1-P7 show sharp head increases throughout these periods of low elevations that, although evident, are of smaller magnitude in the model results. Relatively weak performance is noticeable at 4P and 5P throughout both the calibration and validation periods, and at 8P in the latter. In contrast, despite the issues discussed above, relatively good performance is achieved at the other locations, especially 1P, 2P, 9P and 10P although there is a general over estimation of levels during periods of high head.

Observed and simulated channel stages correspond well on the whole although, with the exception of L2, observations are not as frequent as those for gravel and peat groundwater heads (Figures 4-17 and 4-18). At L2, generally good agreement between observed and simulated river levels in the Lambourn is obtained. Elsewhere, there is a general under prediction of stage during the validation period although, as previously reported, the model performance statistics are generally classified as “very good” to “excellent”. The weakest performance is for W2, especially through the validation period, with the simulated water levels often falling well outside observed stage, underestimating December 2013 to February 2014 and in July 2014, and overestimating through most of the remaining validation period. The model clearly simulates the rapid drops in stage due to the four weed cuts that drive the resulting declines reported at these times in the peat and gravel whilst the increases over the beginning of 2014 are also represented.

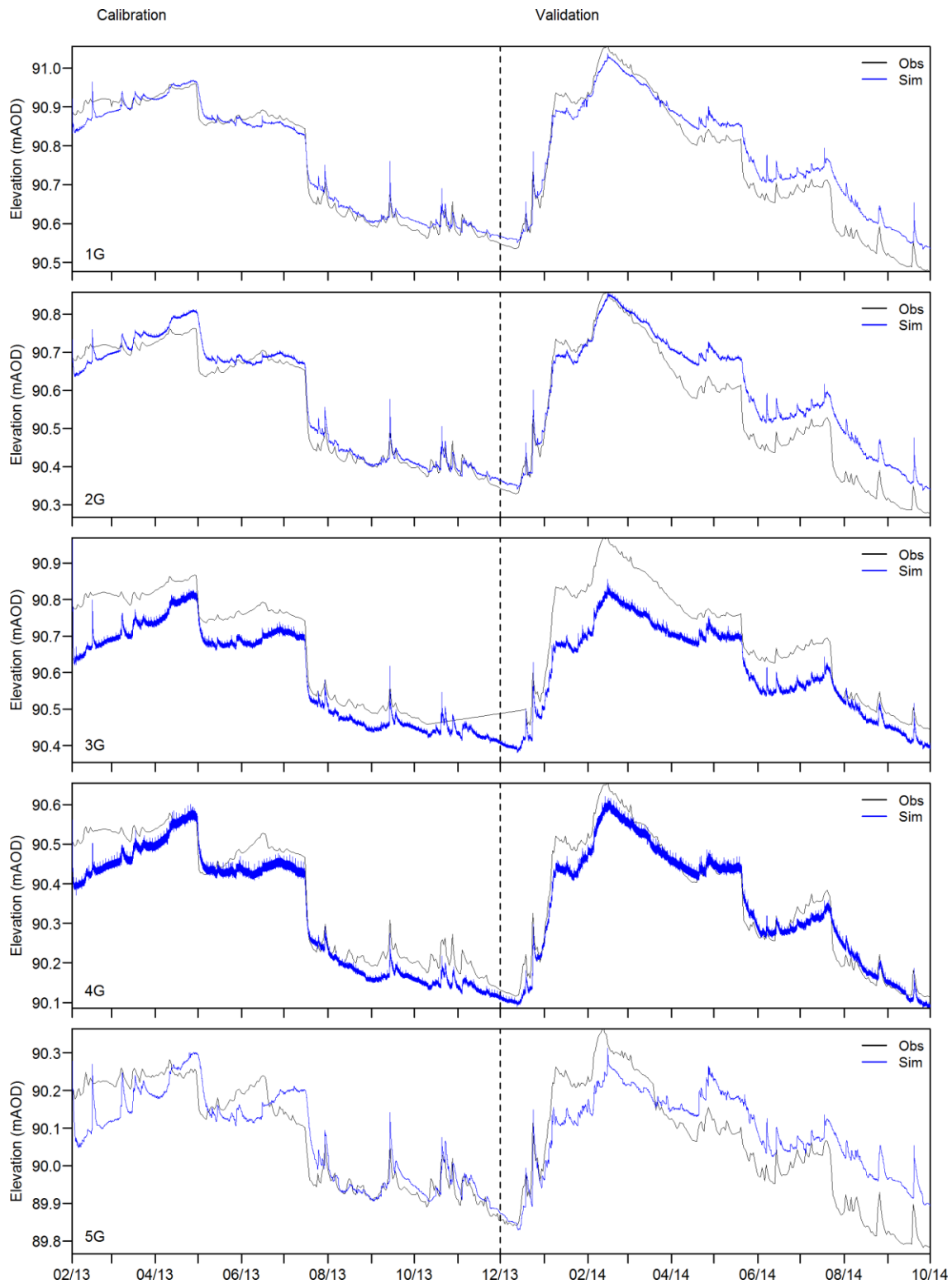


Figure 4-13. Observed and simulated groundwater head elevations (mAOD) in gravel (G) piezometers at locations 1-5

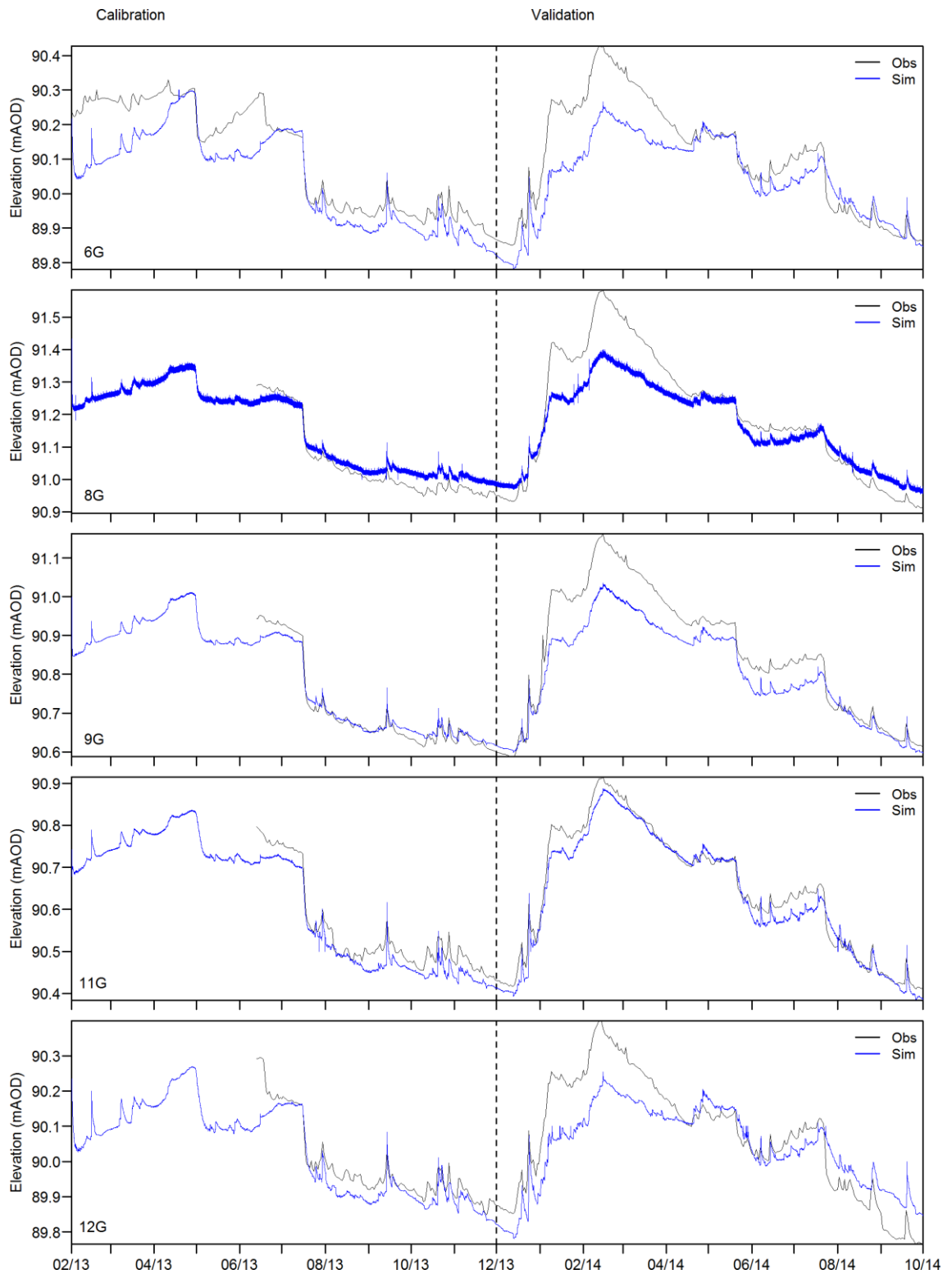


Figure 4-14. Observed and simulated groundwater head elevations (mAOD) in gravel (G) piezometers at locations 6, 8, 9, 11 and 12

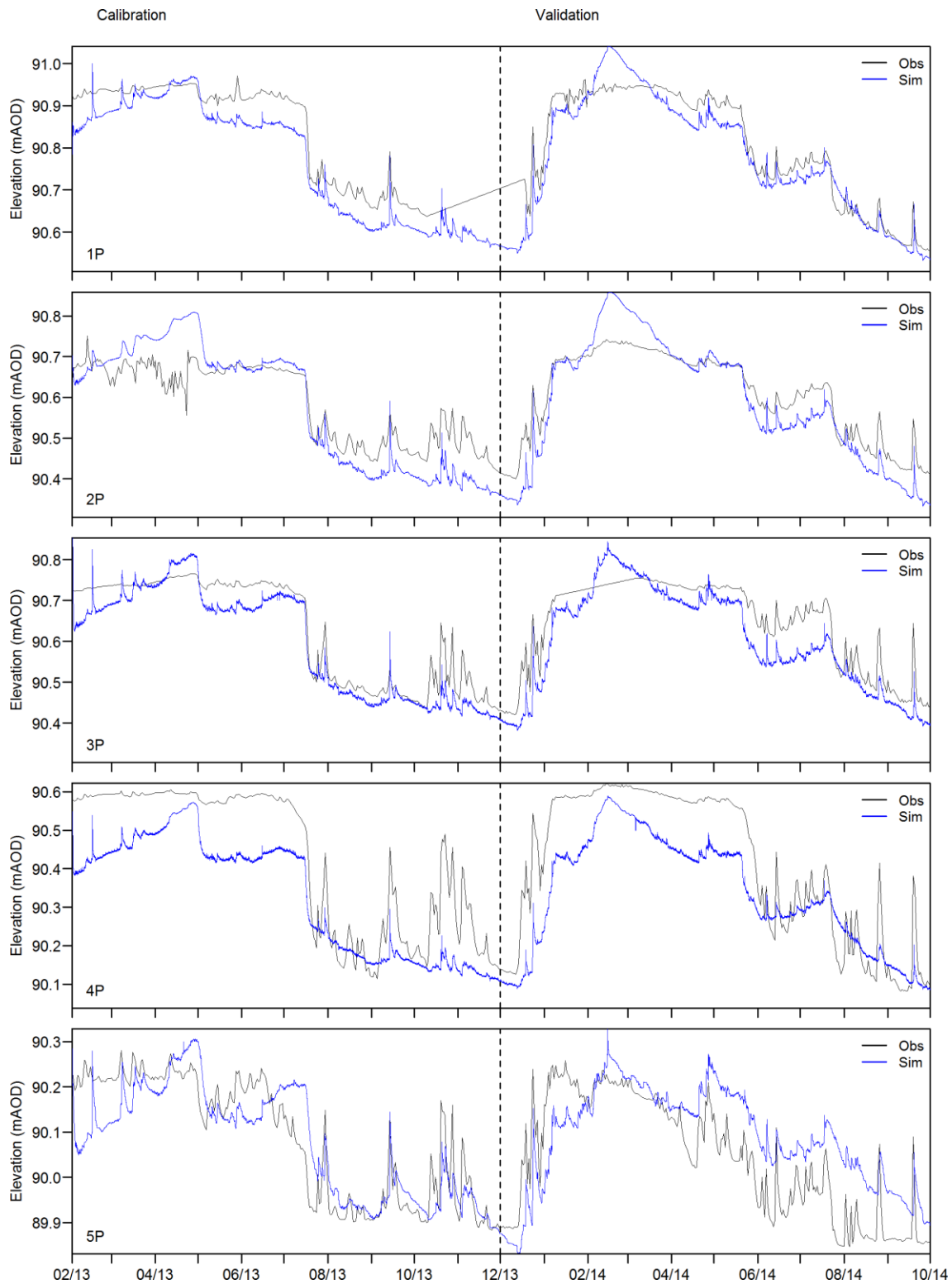


Figure 4-15. Observed and simulated groundwater head elevations (mAOD) in peat (P) piezometers at locations 1-5

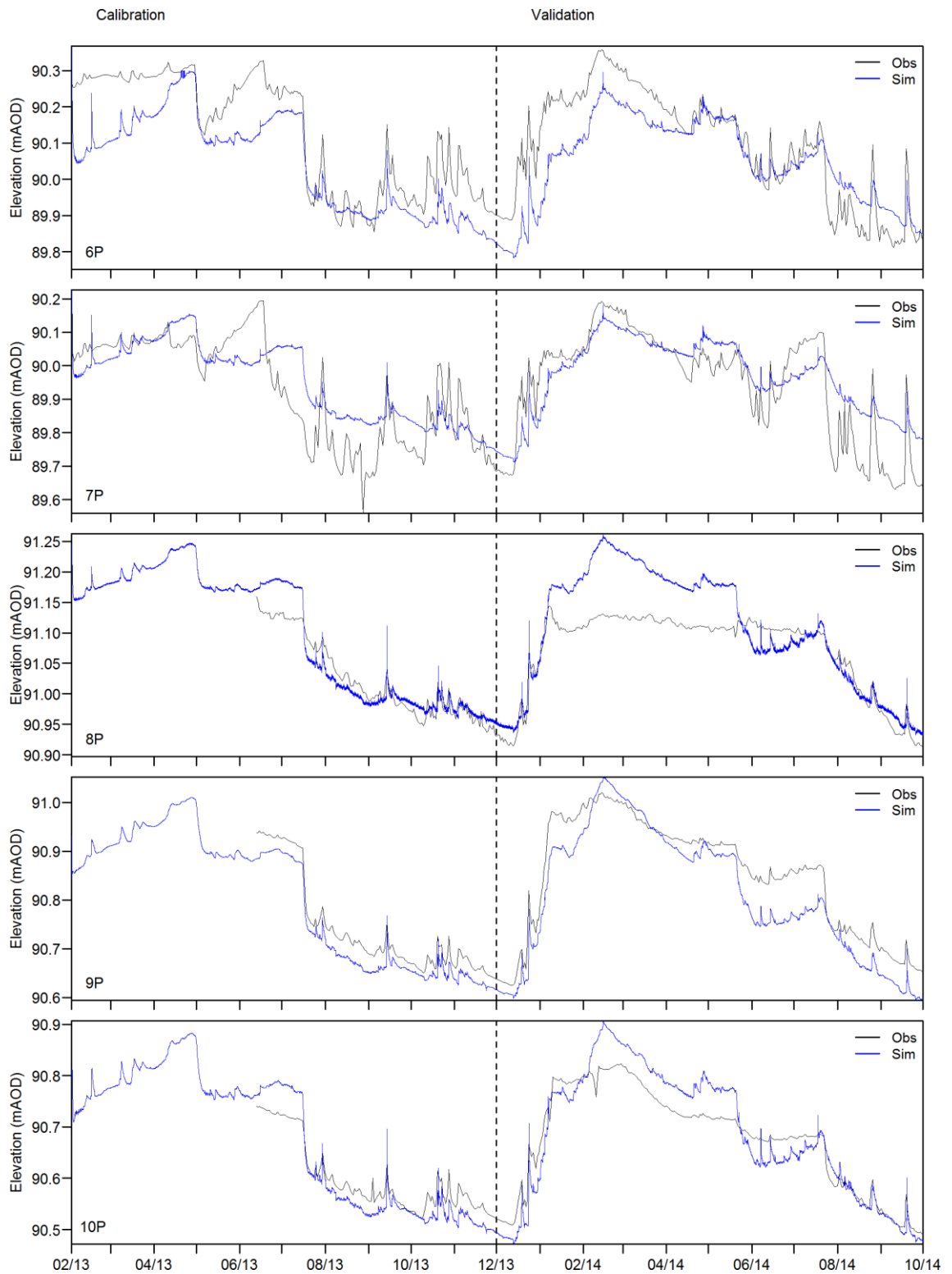


Figure 4-16 Observed and simulated groundwater head elevations (mAOD) in peat (P) piezometers at locations 6-10

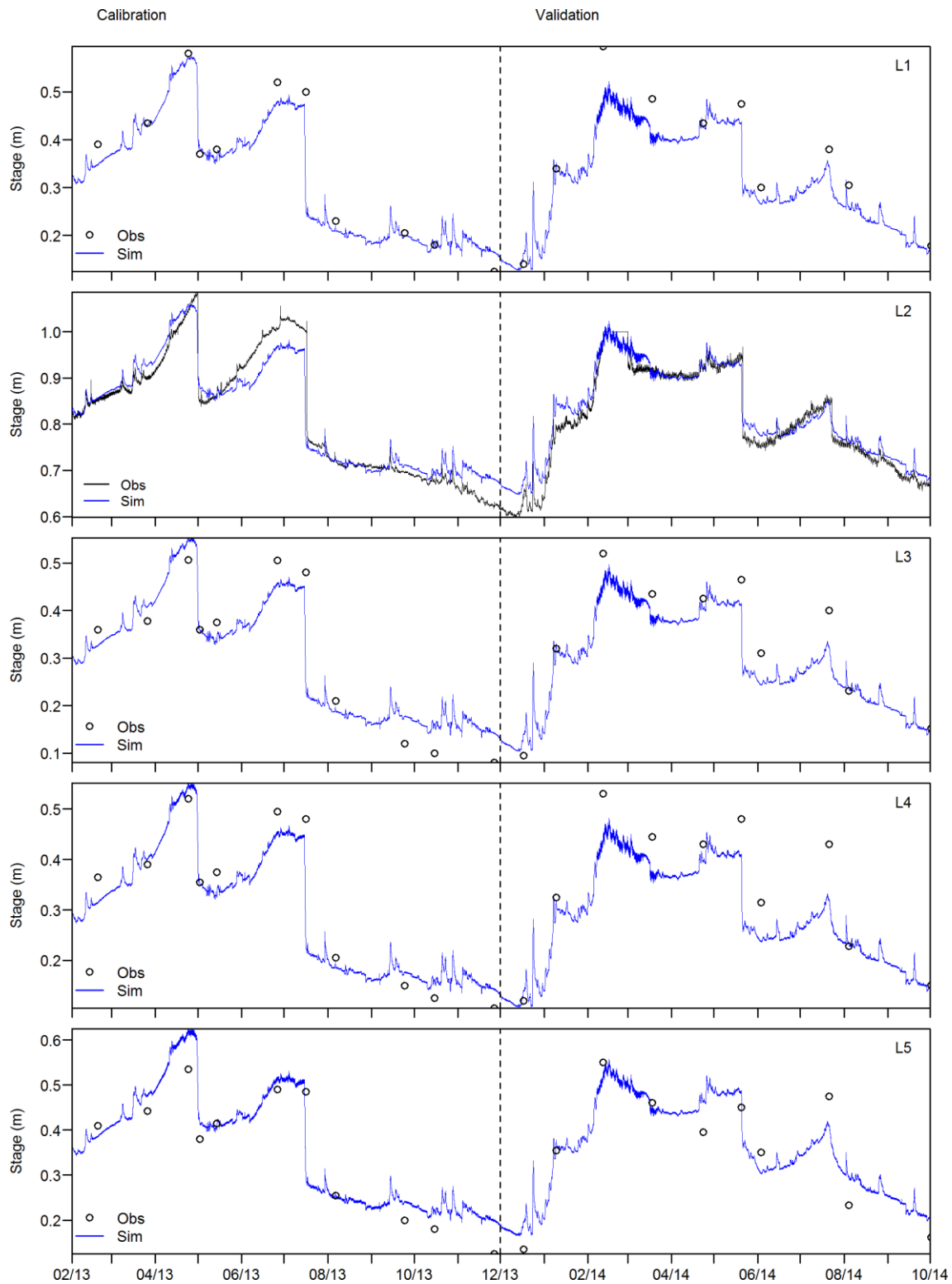


Figure 4-17. Observed and simulated channel stages for the River Lambourn at locations L1-L5

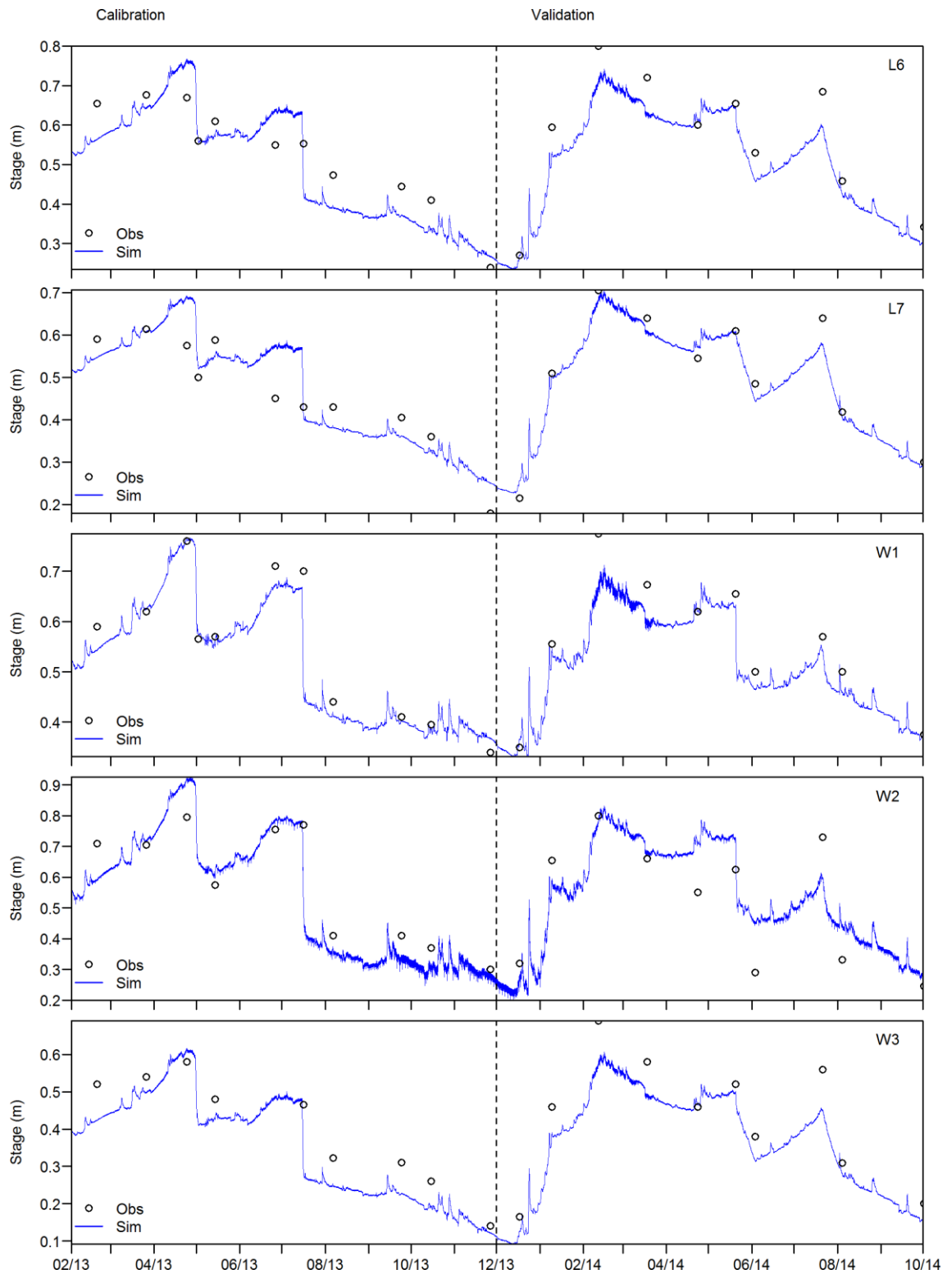


Figure 4-18. Observed and simulated channel stages for the River Lambourn at locations L6, L7, W1-W3

4.4.2 Model performance

Although the model generally simulates conditions very well across the River Lambourn Observatory, there are clearly spatial and temporal disparities in model performance. The superior representation of water levels in particular areas and within the different geological layers highlights the influence of heterogeneity in structure and process at the site scale. Such results also underline the importance of robust field survey and monitoring approaches at spatial resolutions that are sufficient to incorporate this heterogeneity.

Model performance is inferior within the peat. This is especially true when water levels are high and could be due to the inability of MIKE SHE to represent compressible, anisotropic soils, instead defining the hydraulic properties of each geological unit as being temporally and spatially constant. Indeed, incorporation of the effects of soil deformation into hydrological models has, with a few exceptions (Camporese et al., 2006), been generally overlooked. However, peat hydraulic conductivities may vary over relatively short distances by several orders of magnitude (Bragg, 1991; Bromley et al., 2004; Kneale, 1987), seasonally by up to an order of magnitude (Kettridge et al., 2013; Price, 2003), and with depth by several orders of magnitude (Baird et al., 2008; Clymo, 2004). The effectiveness of applying rigid soil theory to peat soils has therefore been questioned (Baird and Gaffney, 1994; Brown and Ingram, 1988). Price (2003) found that saturated hydraulic conductivity was highly correlated to water table depth, with increases of up to 2 orders of magnitude observed after a 0.5 m rise in water table elevation. Peat liquefaction with saturation and the associated increase in hydraulic conductivity could explain the overestimation of head in the MIKE SHE model when water elevations are high, particularly in areas of groundwater upwelling (8P-10P).

Occurrences of silt, sand and gravel within the peat (Allen et al., 2010) could also contribute to local variations in hydraulic conductivity. The presence of these small-scale variations in substrate characteristics are difficult to establish in the field, yet at the applied model grid resolution could have a significant impact on simulated groundwater flow and levels. Variations in the alluvial composition may account for the poorer performance in certain areas, for example at 5P. Additionally, the peat piezometers of the pre-existing array (1P-7P) were installed with the slotted screen extending above ground level (Section 3.2). The sharp observed responses in head during periods of low water levels may, therefore, be a reflection of direct influx of water from the surface during rain events. The exaggerated plateau of high head elevation at 4P may thus be due to surface water above the open level of the piezometer. In contrast, at locations 8P-10P,

where bentonite was used to seal new piezometers with closed screens above ground level, event peak head elevations match well. These instrumental issues could therefore produce misleading results, where otherwise the model could be performing effectively. The influx of surface water into piezometers, though, has not been directly observed and may otherwise be a result of substrate variations and peat compressibility. Differences in magnitude between observed and simulated event peaks vary noticeably by location, while the timings match well. Hence, measurements from these piezometers were not excluded from this study. The model generally simulates gravel head elevations very well. Where deviations occur they fall into two groups; locations where the model underestimates levels (3G, 6G-9G and 12G) and those locations where levels are overestimated towards the end of the simulation period (1G, 2G and 5G). The Electrical Resistivity Tomography (ERT) survey revealed significant braided structures in the gravels (Chambers et al., 2014) (Section 2.5.3). These suggest large differences in gravel porosity across the site which would cause localised and depth dependent variations in hydraulic conductivity, as could quantities of reworked chalk in lower levels (Allen et al., 2010). Although the features could help explain the underestimation at high heads, the over predicted gravel head elevations are more problematical and may be due to inadequacies in the boundary conditions.

Discrepancies between simulated and observed channel stage may in part be due to discrete changes in channel bed roughness. The growth and distribution of instream vegetation is affected by many factors, amongst which channel morphology, bed material and adjacent conditions will contribute (Dar et al., 2014). Spatial variations in species composition, distribution, and abundance due to abiotic variations will cause differences in resistance to flow over potentially small scales. Such localised effects of macrophyte growth on bed roughness are not accounted for within the model; instead, as discussed above, a uniform resistance factor is applied throughout the MIKE 11 river model. The unmonitored sluice gate located just upstream of W2 (Sections 2.2 and 3.5.1) could additionally account for the poor representation of channel stage at this location. Local residents adjust the control structure in order to maintain the aesthetics of a pool feature and the times when the sluice is open or closed are unfortunately not recorded.

4.5 Assessment of hydrological functioning

4.5.1 Water balance

The modelled monthly water balance is summarised so that surface water (SW) represents net outflow (channel and overland outflow minus inflow), while groundwater (GW) represents net inflow (groundwater inflow minus outflow), and baseflow (B) the

exchange between channel and gravels with negative values signifying loss to gravels (Table 4-14). The degree to which surface water and groundwater dominate the water balance is apparent, with SW and GW comprising 44.2 and 43.4 % of total flow (5849.4 mm throughout the whole simulation period) respectively. Precipitation (P) and evapotranspiration (ET) are of secondary importance in transferring water into and out of the site, and constitute 6.3 and 5.7 % of total flow respectively. Groundwater, surface water, precipitation and evapotranspiration are, perhaps unsurprisingly, interlinked, with associated increases and decreases in the first two terms reflecting changes in the balance between the second two. For example, the winter flood period from December 2013 to February 2014 is marked by the increase in water within GW and SW, with both peaking in January 2014, the month with the largest precipitation input. Although storage components are small annually (SWS and GWS), there is obvious temporal variability, with monthly storage changes often as significant as P and ET. Increases in evapotranspiration over the spring and summer periods, when precipitation is relatively low, correspond to periods of reduced water storage.

Selected principal components of the water balance over the full simulation period, with groundwater split by vertical direction and geological layer, are shown in Figure 4-19. The correspondence between increases in surface water flux and rainfall events is clear. In addition the influence of the weed cuts on surface fluxes is clearly demonstrated. The highest peaks occur at the time of the weed cuts in 2013 when rainfall inputs are low or absent, and signify the flush of surface water within the floodplain to channels and in turn to the river in response to the fall in channel stage.

Upward flows of groundwater between layers are consistently higher than downward flows (Figure 4-19). The latter are generally in line with surface water flux, although peak responses are muted in the gravels. However, in the month before each of the weed cuts in 2013, downward flows increase gradually. These increases are mirrored by marked decreases in upward flows, especially between the chalk and gravels. Upward exchanges from the gravels increase sharply with rainfall events and the weed cuts yet follow the general pattern of upward chalk groundwater fluxes. Flow upwards from the chalk displays an inverse pattern, with rapid increases during and immediately after weed cuts, yet decreases in association with rainfall events. Post weed cut flows from the gravels to the peat are maintained at a higher level, corresponding to increased upward flow from the chalk. However, the 23/7/2014 weed cut had comparatively minimal impact. Aside from the conspicuous human induced effects due to the weed cuts, strong seasonality is noticeable, with volumes of groundwater exchanges grading between winter wet periods and summer dry spells.

Table 4-13. Simulated monthly and total water balance for the CEH River Lambourn Observatory. All values in mm. P, precipitation; ET, evapotranspiration; I, interception storage; SW, net surface water outflow; SWS, change in surface water storage; GW, net groundwater inflow; GWS, change in groundwater storage; B, baseflow

Month	P	ET	I	SW	SWS	GW	GWS	B	Error
Feb-13	6.40	9.03	0	136.61	-10.22	145.93	4.17	-0.01	-0.64
Mar-13	26.55	12.25	1.70×10^{-5}	156.72	-7.84	152.50	-2.71	-0.06	0.52
Apr-13	9.60	31.35	-9.42×10^{-6}	53.00	-25.48	106.06	-3.46	0.10	-2.44
May-13	14.65	27.13	-5.52×10^{-6}	172.27	31.85	146.47	5.33	-0.11	1.20
Jun-13	5.75	22.77	0	83.14	-6.27	106.91	-1.89	0.05	1.36
Jul-13	11.25	21.86	-3.20×10^{-5}	118.00	13.76	104.60	9.49	0.07	0.69
Aug-13	5.90	17.67	2.90×10^{-5}	127.78	1.70	133.56	4.50	0.24	-0.46
Sep-13	17.20	8.88	3.73×10^{-5}	123.41	-0.27	115.69	-0.42	0.23	-0.13
Oct-13	31.10	15.44	-2.20×10^{-1}	134.65	-0.14	117.98	0.75	0.17	0.46
Nov-13	15.30	7.06	2.20×10^{-1}	124.49	0.79	112.08	2.40	0.10	0.66
Dec-13	37.45	11.48	-4.59×10^{-2}	158.78	-1.03	138.42	-4.07	-0.20	-0.28
Jan-14	50.70	11.77	-1.08×10^{-1}	210.23	-7.47	189.50	-9.56	-0.27	-0.80
Feb-14	31.95	20.72	1.54×10^{-1}	151.53	-16.54	161.20	-4.77	-0.07	0.46
Mar-14	5.15	18.00	-6.34×10^{-5}	163.00	14.15	158.30	3.44	-0.06	-0.03
Apr-14	24.10	15.88	-5.32×10^{-2}	115.10	-7.03	115.30	-2.15	0.00	0.66
May-14	20.90	21.68	5.32×10^{-2}	130.60	13.49	111.40	7.50	0.00	-1.07
Jun-14	15.50	15.87	-4.97×10^{-2}	122.40	0.63	121.50	-0.14	0.15	0.84
Jul-14	9.10	20.16	4.97×10^{-2}	105.50	1.05	113.50	2.15	0.09	-0.38
Aug-14	21.35	20.97	-1.47×10^{-4}	100.90	1.66	94.80	3.72	0.39	0.03
Sep-14	7.95	4.97	-5.45×10^{-5}	98.20	0.87	89.90	3.21	0.34	0.88
Total	367.86	335.42	1.67×10^{-4}	2589.30	-2.33	2538.80	17.60	1.17	1.61

4.5.2 Surface water flooding and groundwater upwelling

The extent and depth of simulated surface water flooding due to ponding of precipitation and high groundwater table at periods of high flow corresponds closely with topography (Figure 4-20). The shallow relict channels from the historical water meadow system discussed in Section 2.2 are apparent as areas of relatively deep flooding. Elsewhere much of the flooding appears linked to the main channel system. This is especially the case for areas adjacent to the Westbrook as it flows through the centre of the site, and towards the River Lambourn in the southeast. However, in the north meadow and southeast section of the south meadow, areas of flooding not directly linked to the main channel system still occur.

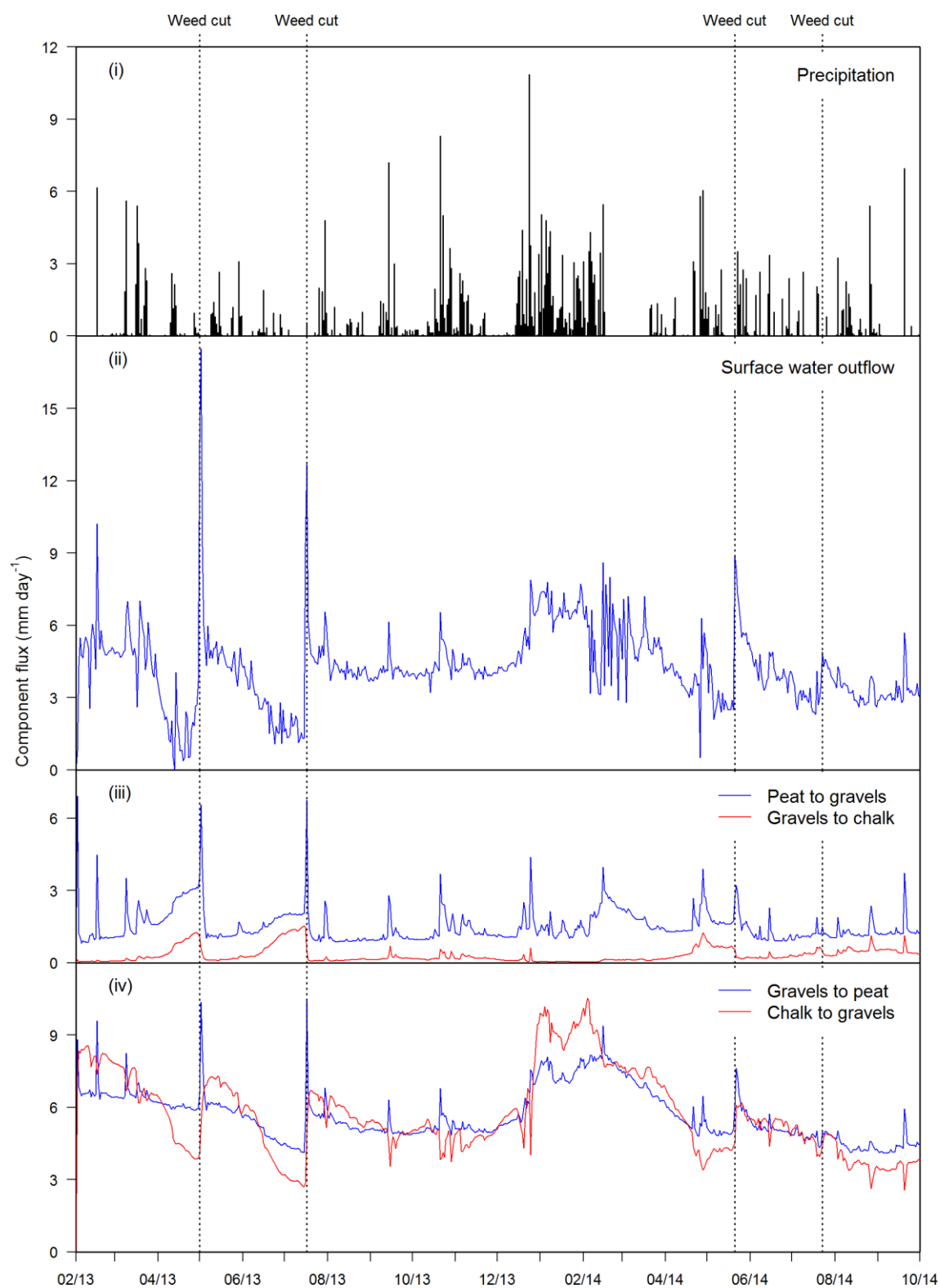


Figure 4-19. Detailed components of the water balance: (i) precipitation, (ii) surface water outflow, (iii) downward groundwater flow between geological layers, and (iv) upward groundwater flow between geological layers

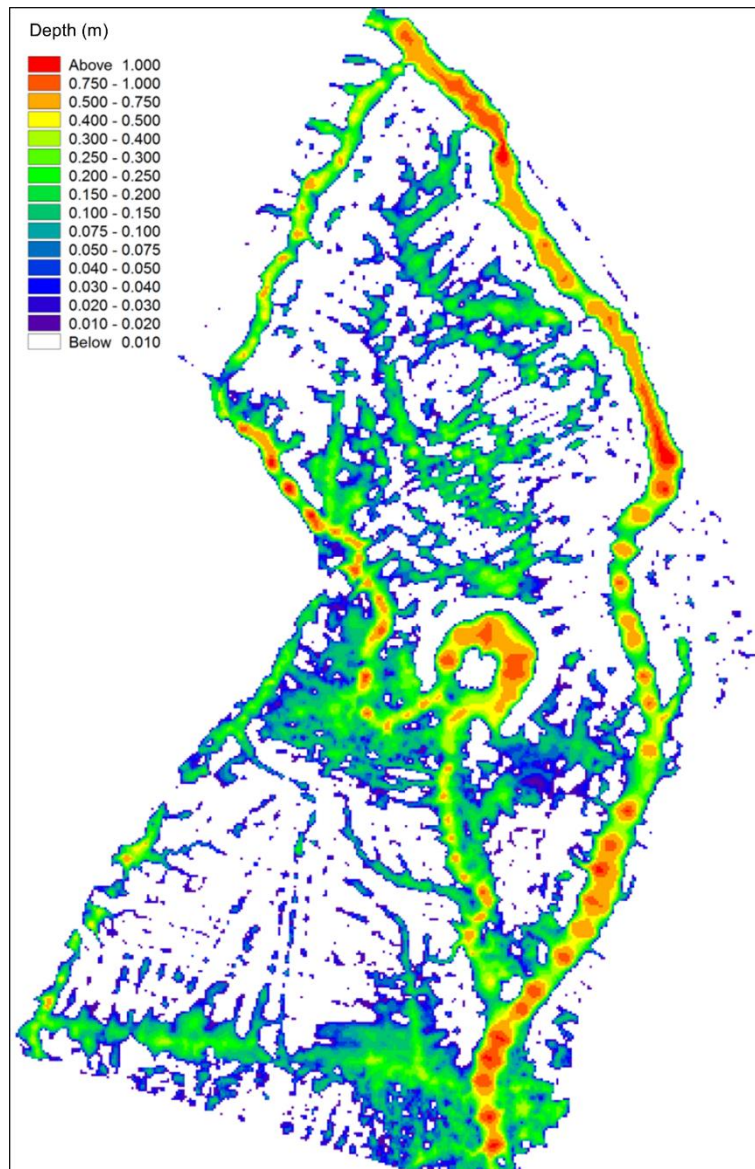


Figure 4-20. Extent and depth of surface water at peak flow (15/2/2014)

Simulated gravel and peat head gradients show an overall resemblance evident in both wet and dry periods (Figure 4-21). Groundwater mounding occurs in both the northern meadow and in the northern part of the south meadow around the Westbrook. A discrete area of particularly high groundwater head is simulated towards the north of the site. These elevated heads in the north meadow correspond with field observations and are concomitant with locations where putty chalk is absent at the interface between the chalk and gravel aquifers. There are additional areas of higher head and hence steeper local head gradients to the centre east, and are especially noticeable in the gravels (Figures 4-21i and 4-21ii). Small-scale head variations in the gravels in line with the Lambourn are evident during the high peak (Figure 4-21ii). Gradients in the peat appear influenced by the topography of the relic drainage network during this peak (Figure 4-21iv), yet less

so at low levels (Figure 4-21iii). In general, head elevations follow the topographic gradient in line with the valley at peak elevations (Figure 4-21ii and 4-21iv). However, there is a shift in the direction of groundwater flow from towards the south to the southwest as head elevations drop (Figure 4-21i and 4-21iii).

4.5.3 Groundwater/surface water interaction

Surface water and groundwater are inextricably linked and crucial to processes in the wetland. The channels, gravels and peat are hydraulically connected. Model results show that gravel waters provide a significant contribution to the site, supporting earlier findings of research in the Lambourn (Abesser et al., 2008; Grapes et al., 2006). Channel stage acts as a head boundary and controls broad water levels within both the gravels and peats. It also influences responses in groundwater flow from the Chalk aquifer. Chalk groundwater is an important source of water into the Lambourn Observatory, discharging into the gravel aquifer and then the wetland through gaps in the putty chalk and resulting in locally elevated heads. Rapid reductions in head elevation immediately after weed cutting in the River Lambourn draw water up from the chalk, increasing the rate of upward groundwater flow. Conversely, increased stage resulting from storm events raises head elevations within the gravels and peats, inhibiting upwelling from the chalk groundwater. The influx of surface water into the gravels at high stage drives the increases in gravel head.

The longer-term trend of surface water outflow follows the seasonal pattern of groundwater inflow. When heads in the Chalk are high, so are levels in all components of the system. This reflects larger-scale catchment processes and the position of the site in a chalk valley bottom with a groundwater fed river. Surface flooding is a combination of seepage from upwelling groundwater, and overbank flow routed from the channels by the relic drainage network. The simulated areas of groundwater mounding in the north meadow and associated flooding support earlier findings in the field (Section 3.7.2). To the east, steeper head gradients correspond with the mouth of a dry valley. However, the cause of the high heads around the Westbrook is less clear. This area does, however, fall beyond the extents of the detailed topographical and geological surveys, with access limited by dense vegetation. It is difficult to assess whether the results are from a real or interpolated feature, and this highlights the importance of high-resolution field data.

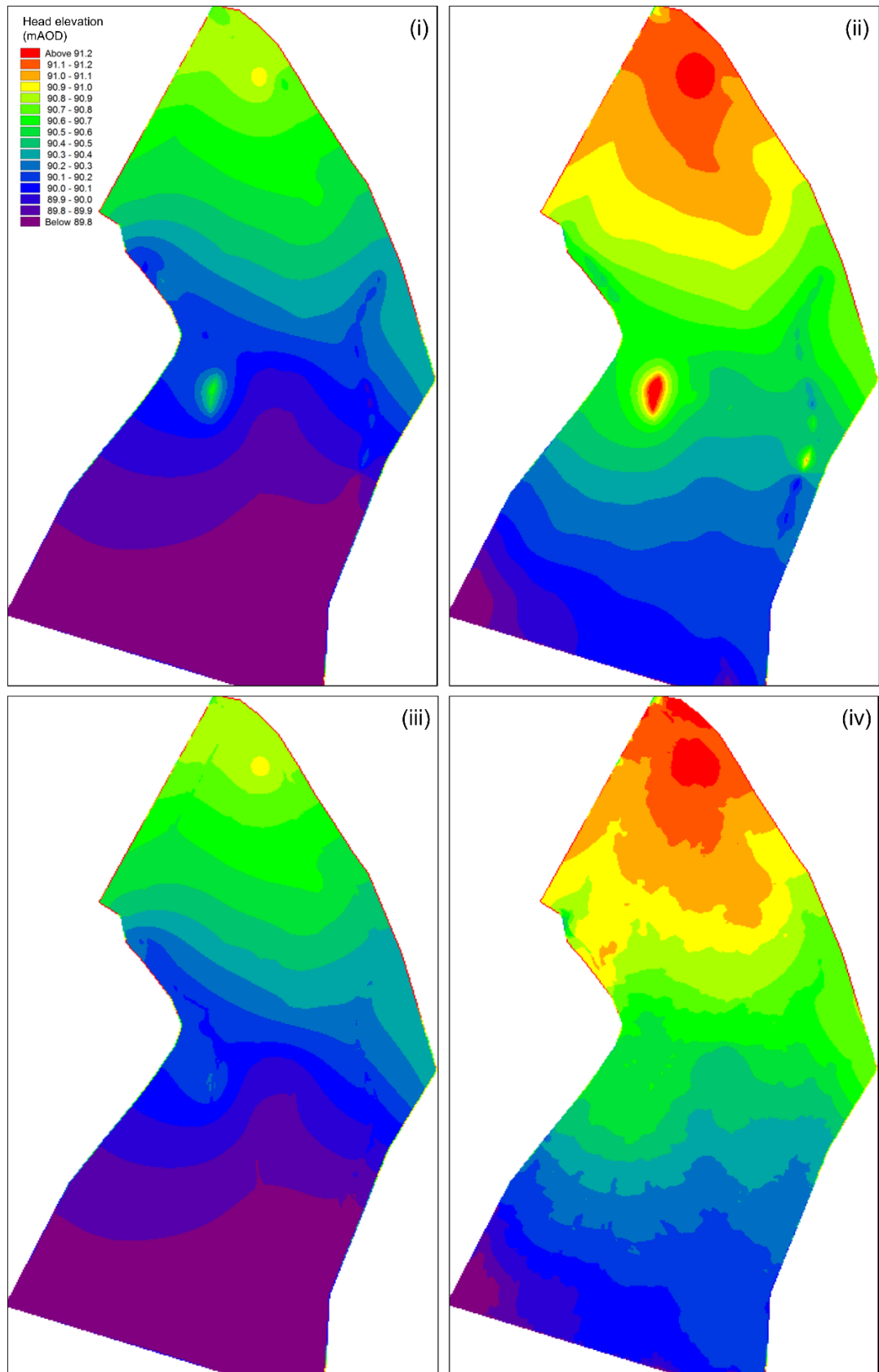


Figure 4-21. Head elevations in (i) gravels at the lowest head elevation (12/12/2013), (ii) gravels at the highest head elevation (15/2/2014), (iii) peat at the lowest head elevation (12/12/2013 and (iv) peat at the highest head elevation (12/2/2014)

4.6 Summary

A number of models have been reviewed for simulation of wetland hydrology at the River Lambourn Observatory, where adequate representation of surface-groundwater interactions is an essential requirement. Although many available models were ruled out on the basis of this requirement, the commercial and academic sector yielded five that were considered suitable for appraisal. These include HGS, MIKE SHE, MODHMS, SHETRAN and WaSIM.

Selection criteria were developed from the existing conceptual understanding at the Observatory and inherent identification of the principal hydrologic processes, along with an evaluation of previous applications to wetland modelling. These criteria allowed ratings to be applied against various aspects of model capability. The ratings were reviewed and scored in order to identify the appropriate model. Models were classified on a matrix of functionality, adaptability and accessibility to aid the decision making process.

The MIKE SHE modelling system was selected for use in this study. An ability to adapt the complexity of process representation to the applications, a modular framework, and the calculation of each process at distinct and appropriate time steps and spatial scales distinguished MIKE SHE from the other relevant modelling systems. Such flexibility, in addition to a proven applicability for simulating wetland hydrology, underpins its use in this study.

The potential of MIKE SHE to model wetlands with a complex subsurface architecture has been demonstrated for the River Lambourn Observatory. Findings support the conceptual model, with hydrological processes in the wetland dominated by the interaction between groundwater and surface water. Channel stage provides head boundaries for broad water levels across the wetland, whilst areas of upwelling from the Chalk aquifer control discrete head elevations. A relic surface drainage network confines flooding extents and routes seepage to the main channels.

Model performance is generally very good. Results are consistent with field observations and follow short-term responses to hydrological and management events, as well as the longer-term seasonal cycle. The impact of instream weed cutting is well represented, and affects water levels throughout the site. The interaction between surface water and groundwater is also markedly affected by weed cutting. This influences head variations across the wetland, the proportional contribution of each water source, and the maintenance of areas of standing water.

The MIKE SHE model of the Lambourn Observatory has the potential for investigating the hydrological effects of environmental changes, whether from alterations to climate, groundwater abstraction, channel morphology or vegetation management. Representative boundary conditions could be obtained through links to existing regional groundwater models (e.g. Jackson et al., 2011), although differences in model grid resolution would need to be addressed. Further application of ecological indices to model results will allow assessment of the ecological sensitivity of the wetland to environmental changes (e.g. Thompson et al., 2009). Specific water level requirements of plants and animals in the wetland, and environmental flows in the channels, could be linked to species maintenance or succession. The MIKE SHE model of the Lambourn Observatory therefore represents an essential tool for understanding the wetland ecosystem and its response to change and for developing management approaches. These applications of the model are investigated in the following chapters.

Chapter 5

Hydrological effects of climate change on the River Lambourn Observatory

5.1 Introduction

Projected changes in climate are likely to substantially impact wetland hydrological conditions (Baker et al., 2009; Dawson et al., 2003; Erwin, 2009). The effects of climate change on regional aquifers and catchment runoff may cause intricate and significantly detrimental impacts to wetlands underlain by permeable geology, such as the chalk lowlands of southeast UK (Herrera-Pantoja et al., 2012) where the Observatory is located. The impacts of climate change upon such wetlands should ideally therefore be assessed on an individual basis in relation to their water supply mechanisms and position within the catchment (Acreman et al., 2007). Changes in water table level of less than 0.1 m may have profound effects on species composition, and provide conditions which favour distinct species or communities over those currently dominant at a given site (Wheeler et al., 2004). There are relatively few hydrological modelling studies at a suitable resolution which link water table predictions directly to plant and animal requirements for individual wetlands (Carroll et al., 2015; Thompson et al., 2009). To the author's knowledge, none do so for individual wetlands with groundwater contributions. However, an ability to accurately predict the impacts of climate change is vital for wetland management where species conservation and ecosystem service provision relies on managing hydrological functions (Acreman et al., 2009). Models able to accurately represent wetland hydrology will enable the assessment of possible degradation to wetland ecosystems through climate change (Acreman and Jose, 2000).

In this chapter, the impacts of climate change on the hydrological functioning of the Lambourn Observatory are assessed. Projected changes in hydrometeorological inputs to the MIKE SHE hydrological/hydraulic model of the wetland, described in Chapter 4, were derived from the UK Climate Projections 2009 ensemble of climate models for the 2080s under different scenarios. These comprise scenarios of different climate sensitivities to incorporate the uncertainty associated with climate change. The hydrological model is used to investigate how climate change scenarios affect hydrology in the Observatory.

5.2 Climate change scenarios

Climate change scenarios were derived for the 2080s using datasets from the Future Flows and Groundwater Levels project (Jackson et al., 2011; Prudhomme et al., 2012). These include 11-member ensembles of 1 km gridded time series projections (1950–2098) of precipitation, PET, and groundwater levels for Great Britain based on the UKCP09 Hadley Centre’s HadRM3-PPE run under the medium emissions (SRES A1B) scenario (Murphy et al., 2009). The Met Office Hadley Centre’s Regional Climate Model HadRM3 represents parameter uncertainty through model variants with different climate sensitivity, defined as the equilibrium mean surface temperature change resulting from a doubling of the atmospheric CO₂ concentration (IPCC, 2014). The ensemble was designed to sample the range of uncertainty associated with the parameters of the HadRM3 atmosphere. However, the ensemble under samples GCM uncertainty and excludes emissions scenario uncertainty.

HadRM3-PPE consists of an ensemble of eleven members of HadRM3 used to dynamically downscale HadGM3 global climate model outputs (Murphy et al., 2009). The ensemble comprises one unperturbed member and 10 members with different perturbations to the atmospheric parameterisations (HCCPR, , 2008). Climate sensitivities for each ensemble member along with the scenario run id plus the RCM run id and descriptive id used by the Met Office Hadley Centre are summarised in Table 5-1. Three scenarios (H, J and K) have climate sensitivities above the likely range of 2-4.5 °C estimated by the IPCC (IPCC, 2014). Outputs from HadRM3-PPE are provided at a 25 km grid resolution. Due to differences in scale between local hydrological processes and modelled atmospheric processes from the RCM, a bias correction and spatial downscaling procedure was applied to these outputs to obtain the Future Flows precipitation and PET projections (Prudhomme et al., 2012). PET time series were calculated using the Penman-Monteith equation using projected values of the equation’s meteorological components.

A British Geological Survey (BGS) ZOOMQ3D regional groundwater model of the Chalk aquifer of the Marlborough and Berkshire Downs and south-west Chilterns (Jackson et al., 2011; Figure 5-1) was used to provide the Future Flows projections of changes in groundwater levels (Haxton et al., 2012). It was not possible to drive the entire chalk boundary of the MIKE SHE model of the Lambourn Observatory with predictions from the regional model as the grid was too coarse at 500 m × 500 m.

Table 5-1. Climate sensitivities, scenario ID and model variant name for the HadRM3-PPE ensemble of climate projections (after HCCPR, , 2008)

Scenario ID	Climate sensitivity	RCM run ID	RCM name
A	3.53485	afgcx	HadRM3Q0
B	2.58475	afixa	HadRM3Q3
C	2.81543	afixc	HadRM3Q4
D	3.43839	afixh	HadRM3Q6
E	4.39594	afixi	HadRM3Q9
F	3.89523	afixj	HadRM3Q8
G	4.44284	afixk	HadRM3Qk
H	4.88248	afixl	HadRM3Q14
I	4.54486	afixm	HadRM3Q11
J	4.79648	afixo	HadRM3Q13
K	7.11014	afixq	HadRM3Q16

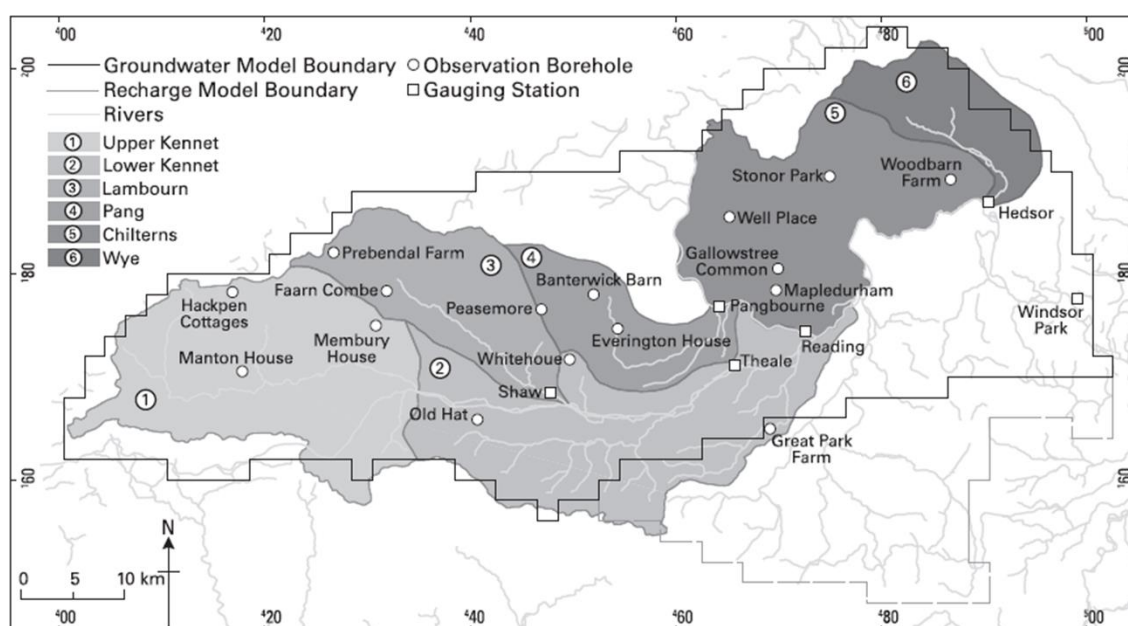


Figure 5-1. ZOOMQ3D regional groundwater model domain and catchments for the Chalk aquifer of the Marlborough and Berkshire Downs and south-west Chilterns (Jackson et al., 2011)

5.3 Climate change impacts on model hydrometeorological inputs

5.3.1 Perturbation of the MIKE SHE model with delta factors

For investigation of the effects of climate change, the MIKE SHE model area was discretised using a 5 m × 5 m grid, producing 4261 computational cells. The computational time for each model run was approximately 30 minutes. Model inputs of precipitation, PET, groundwater elevation and river discharge were perturbed for each climate change scenario using a delta factor approach (Thompson, 2012; Wilby and Harris, 2006). The baseline simulation comprised the combined calibration and validation period 01 Feb 2013–01 Oct 2014. Although this is a relatively short period, constraints were imposed by data availability and the approach replicates those used elsewhere (e.g. Thompson et al., 2009). Monthly percentage differences between the ensemble reference period (1961–1990) and the future period (2071–2098) were applied to each variable. This approach assumes that climate variability does not alter and provides no information on changes in event frequency and distribution (Chiew et al., 1995; Graham et al., 2007). However, it enables a robust comparison of average outcomes and has been widely used in hydrological studies of climate change (e.g. Arnell, 2004; Arnell and Reynard, 1996; Jackson et al., 2011; Kamga, 2001; Limbrick et al., 2000; Thompson et al., 2009).

Monthly delta factors for precipitation (%), PET (%) and groundwater level (m) were extracted from the relevant 1 km grid square of the Future Flows dataset for the HadRM3 ensemble. In the absence of extant delta factors for discharge for the study location from the Future Flows dataset, a rainfall-runoff model was developed for the Lambourn catchment at Shaw. This was developed using MIKE NAM, a deterministic, lumped model describing, in a simplified quantitative form, the behaviour of the land phase of the hydrological cycle (DHI, 2009). Following model calibration, climate change delta factors for discharge were derived by running the NAM model with catchment averaged precipitation and PET under each of the 11 HadRM3 ensemble members. These factors, expressed as a percentage, were subsequently applied to the original stream inflows used within the MIKE SHE model that were based on the relationship between discharge immediately upstream of the model area and at the Shaw gauging station.

Daily precipitation for the NAM model of the 234.1 km² Lambourn catchment was obtained from the CEH-GEAR dataset (Keller et al., 2015) which provides 1 km gridded estimates of daily and monthly rainfall for Great Britain and Northern Ireland derived from the Met Office national database of observed precipitation. Monthly PET totals (subsequently disaggregated to a daily time step assuming an even distribution through

the month) were taken from the Met Office Rainfall and Evaporation Calculation System (MORECS), which is based on the Penman-Monteith equation and provides UK-wide coverage at a 40 km² grid square resolution (Thompson et al., 1981). Spatially uniform time series of both precipitation and PET were derived from the mean of those cells for the two datasets falling within the catchment. Calibration and validation of the NAM model was based on comparisons between daily observed and simulated discharge at the Shaw gauging station for the equally split period 1/1/1963 – 30/6/1987 and 1/7/1987 – 31/12/2012 (Figure 5-2). An automatic multiple objective calibration routine was based on agreement between mean simulated and observed runoff along with the root mean square error. Adjusted parameters included maximum water content in the surface and root zone storage, the overland flow runoff coefficient, time constants for interflow, routing overland flow and routing baseflow, and the root zone threshold values for overland flow, interflow and groundwater recharge. As with the MIKE SHE model of the Observatory, performance was classified following the Henriksen et al. (2008) scheme. RMSE in this case was deemed excellent if below 0.5 m³s⁻¹. Performance was better through the validation period. Calibrated values for RMSE, R and R² were 0.435 m³s⁻¹, 0.81 and 0.66 respectively. Validated values for RMSE, R and R² were 0.399 m³s⁻¹, 0.92 and 0.84 respectively. This is likely due to a model warm up period up to approximately 1965, visible in Figure 5-2 where observed and simulated values are not in agreement.

5.3.2 Climate change impacts on model hydrometeorological inputs

Monthly delta factors for the hydrometeorological time series used to drive the MIKE SHE model of the Lambourn Observatory are summarised in Figure 5-3 for each of the 11 ensemble member scenarios as well as the scenario mean. Drier summer and wetter winter months are evident from the precipitation and PET change factors although the magnitude and duration of changes vary between scenarios (Figure 5-3i and 5-3ii). The scenario mean shows increases in precipitation between October and March and decreases during the months April–September (Figure 5-3i). November contains the largest increase and greatest range of changes (+2.6 % for scenario E to 61.6 % for B) while the largest projected mean decline is in August (-38.3 %). Delta factors for PET are positive in every month for all of the scenarios (Figure 5-3ii). They are largest in late summer and smallest in mid to late winter and consequently the scenario mean ranges from +56.4 % in September to +19.6 % in March. The maximum individual increase (+75.6 % for scenario H) occurs in September whilst the minimum increase

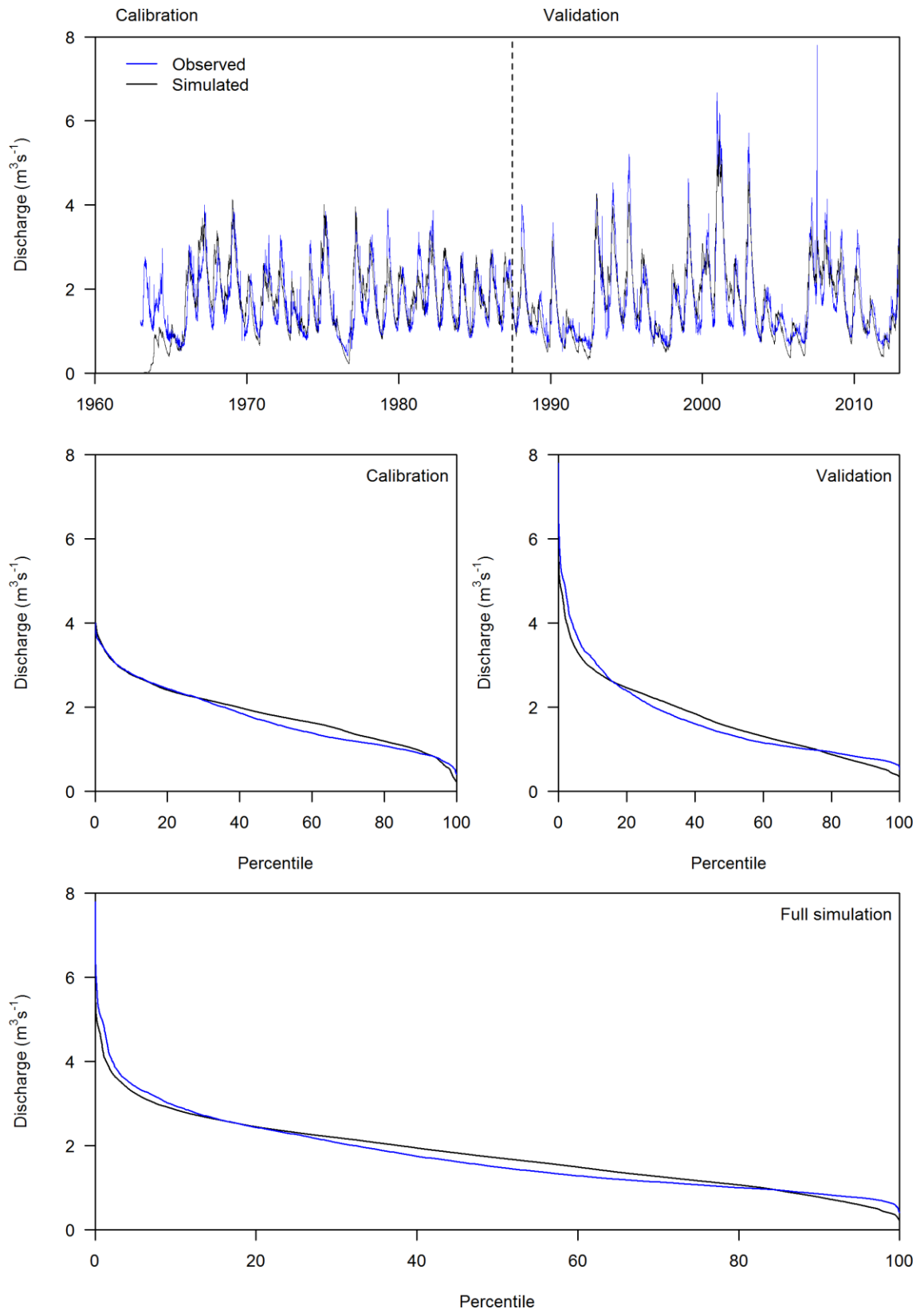


Figure 5-2. Observed and simulated discharge of the Lambourn catchment at Shaw gauging station for the simulation period 1963-2012, and flow duration curves for the calibration (1/1/1963 – 30/6/1987), validation (1/7/1987 – 31/12/2012) and full simulation periods

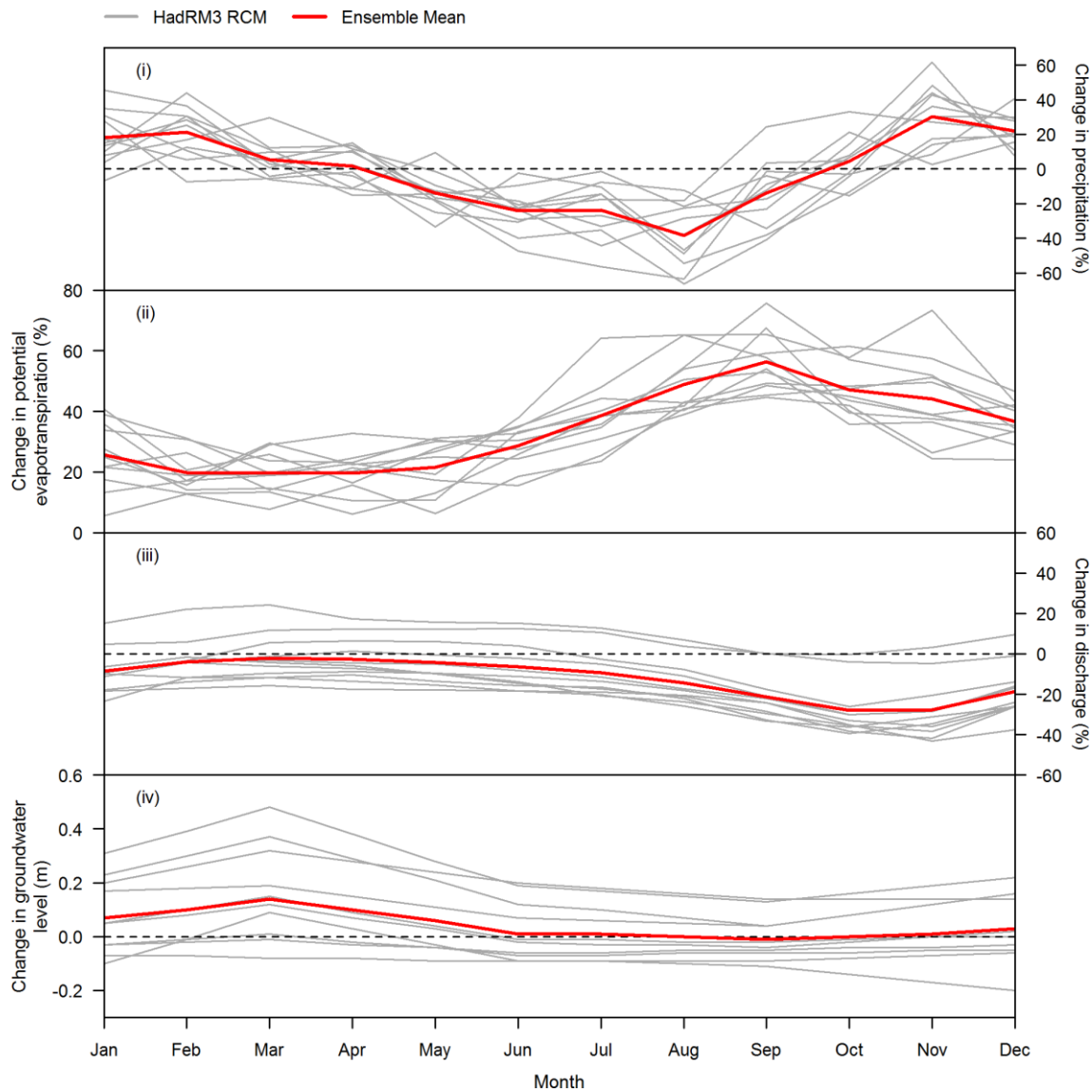


Figure 5-3. Projected monthly climatic changes for the 2080s by scenario and mean: (i) precipitation, (ii) potential evapotranspiration, (iii) river discharge, and (iv) groundwater level

(+5.6 % for E) is projected for January. The inter-scenario range is particularly large in the latter half of the year when delta factors are the largest.

The discharge delta factors for the scenario mean suggest declines in river flow throughout the year (Figure 5-3iii). These are largest in October, whilst the declines are smallest in March (-2.0 %). Of the 11 individual scenarios only three show increasing discharge at any time of the year. The remaining eight scenarios project declines in discharge throughout the year. The scenario mean delta factors for Chalk aquifer groundwater levels generally show an increase, especially over the late winter months (Figure 5-3iv). Exceptions to these increases occur in August and October when there is no change, and September when small declines (-0.01 m) in groundwater level are

projected. Only four scenarios project year round declines in groundwater levels, and four scenarios project year round increases, though the remainder show declines from late spring, through the summer and into the autumn/winter.

The effects of the delta factors on total annual baseline precipitation and PET as well as mean annual discharge and groundwater level for the complete hydrological year (01 Oct 2013-30 Sep 2014) of the simulation period are displayed in Table 5-2. The scenarios can be divided into two groups: those with net precipitation ($P - PET$) above 100 mm which correspond to A, B, C, D and E with a mean climate sensitivity of 3.35 °C; and, those with net precipitation below 100 mm (F, G, H, I, J and K with a mean climate sensitivity of 4.95 °C). The former are characterised by relatively larger increases in precipitation and groundwater level, smaller increases in PET and either increases or decreases in mean discharge. Members of the second group have, on the whole, smaller increases (declines for I) in precipitation and larger increases in PET. Mean discharge and groundwater level tend to decline although some individual members provide exceptions to these general trends. Total inflows to boundary conditions are also shown in Table 5-2. Percentage changes under each scenario are within the same order of magnitude as changes in precipitation. A multiple regression comparison with precipitation and PET yields a good relationship ($R^2 = 0.72$).

5.4 Climate change impacts on hydrology

5.4.1 Wetland water levels

Climate change related modification to wetland water levels varies spatially and temporally (Figures 5-4, 5-5 and 5-6). Water level responses fall into three spatial groups: locations in the north meadow that are characterised by upwelling groundwater (North – Upwelling; Figure 5-4), locations in the same part of the wetland where such upwelling is absent (North – no upwelling; Figure 5-5), and locations in the south of the wetland (South; Figure 5-6) where inter-scenario variability is small in comparison to northern parts of the wetland, especially during periods of relatively high water. Hence, Figures 5-4, 5-6 and 5-5 show simulated wetland water levels at locations that are characteristic of these groups (piezometers 1-4 in Figure 5-4; piezometers 8-9 in Figure 5-5; piezometers 5-7 in Figure 5-6), for the baseline scenario, each of the 11 scenarios and the scenario mean.

Table 5-2. Annual baseline precipitation (P) (mm), potential evapotranspiration (PET) (mm), precipitation minus PET (P-PET) (mm), mean discharge (Q) (m^3s^{-1}), mean chalk groundwater head at 3C (G) (mBGL) and total boundary inflow (mm) and changes (% for precipitation and discharge and inflow, m for groundwater head, positive upward) in 2013/2014 for scenarios and mean. Italicised values indicate negative changes

Run ID	P (mm/%)	PET (mm/%)	P-PET (mm)	Q (m^3s^{-1} /%)	G (mBGL/m)	Inflow (mm/%)
baseline	1081.4	764.3	317.1	2.37	0.31	20365.9
A	6.6	31.6	146.5	-6.53	0.11	5.2
B	2.8	26.8	142.6	-18.68	-0.08	-0.8
C	5.9	25.4	186.2	8.04	0.20	7.9
D	15.9	26.7	285.3	17.49	0.25	10.3
E	7.8	24.4	214.9	-2.48	0.17	8.0
F	0.8	35.1	57.3	-13.08	0.02	2.4
G	1.2	36.2	53.6	-11.32	-0.08	-0.7
H	3.4	36.8	72.3	-16.23	-0.04	0.3
I	-2.8	35.5	15.8	-20.38	-0.04	0.6
J	5.5	38.8	79.4	-8.91	0.03	2.5
K	5.1	38.1	80.6	-16.77	-0.08	-1.3
mean	4.7	32.3	121.3	-8.08	0.04	2.8

The largest inter-scenario range in simulated levels occurs towards the end of October 2013 and corresponds to low flow conditions whilst the smallest range corresponds to the high flows period of February 2014. Both non-upwelling and upwelling locations in the north meadow have relatively large inter-scenario ranges, varying at an individual location between 0.05 m (location 9) and 0.31 m (locations 1, 2 and 4). In the south the range is smaller, varying between 0.04 m (location 5) and 0.19 m (location 7). Changes in water levels for the scenario mean are relatively small with projected water levels being close to the baseline throughout the simulation period. To illustrate, over the full simulation period the mean difference between the baseline and the scenario mean is 0.00 m in the north, while in the south mean differences suggest a decline of -0.03 m (Table 5-3).

In the south meadow, the absence of periods when the water level exceeds the ground level under baseline conditions is repeated for each climate change scenario. However, in the north meadow, some scenarios result in water levels rising above the ground when under baseline conditions this did not occur (e.g. location 3, scenarios C, D). Elsewhere

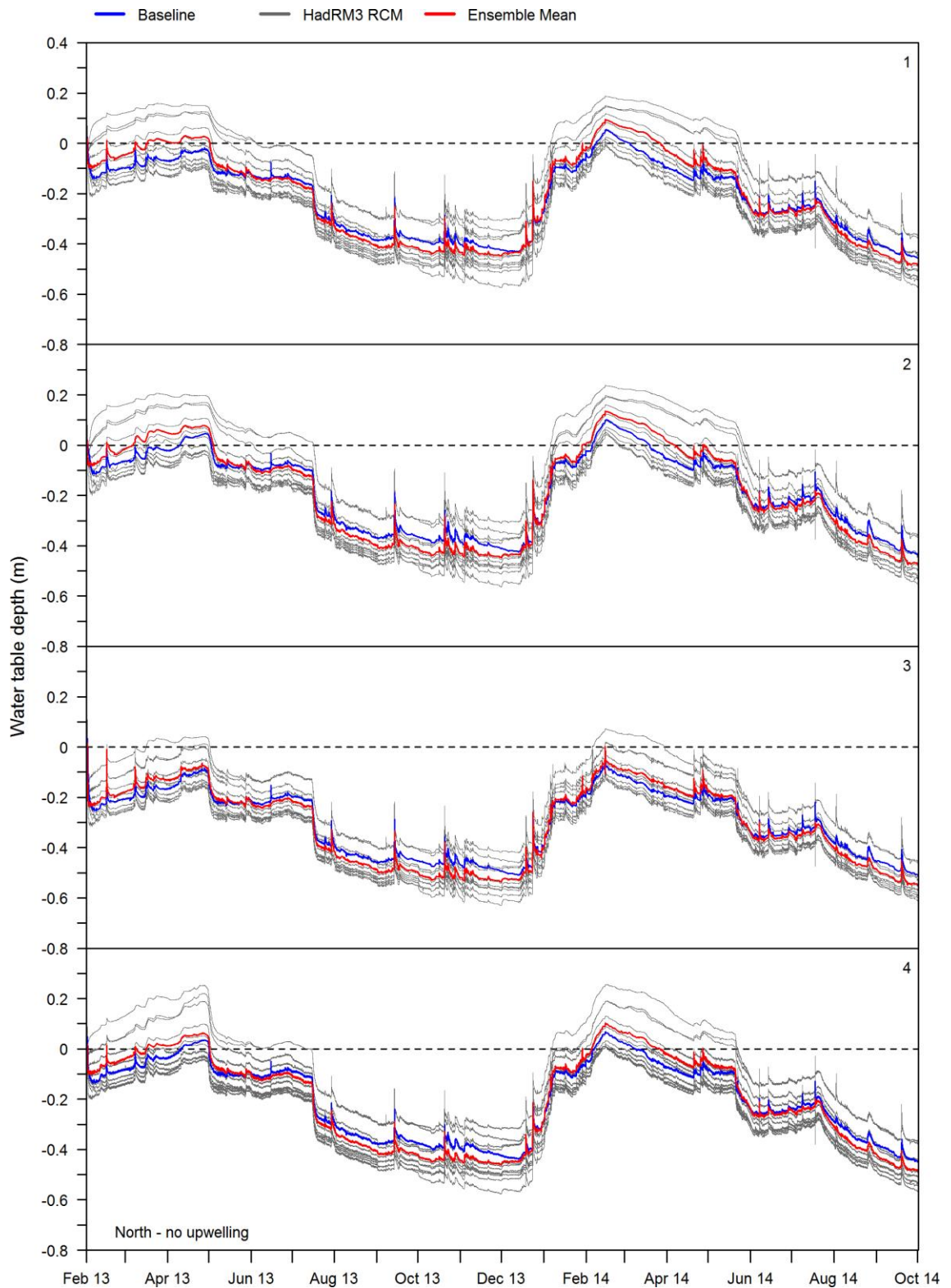


Figure 5-4. Simulated baseline, projected scenario and mean wetland water table depths for North no upwelling locations

where baseline water levels did exceed ground level, some scenarios increase the depth and duration of groundwater induced surface flooding. For example, for some scenarios (A, C, D and E) surface water levels are increased by up to 0.2 m and the period of

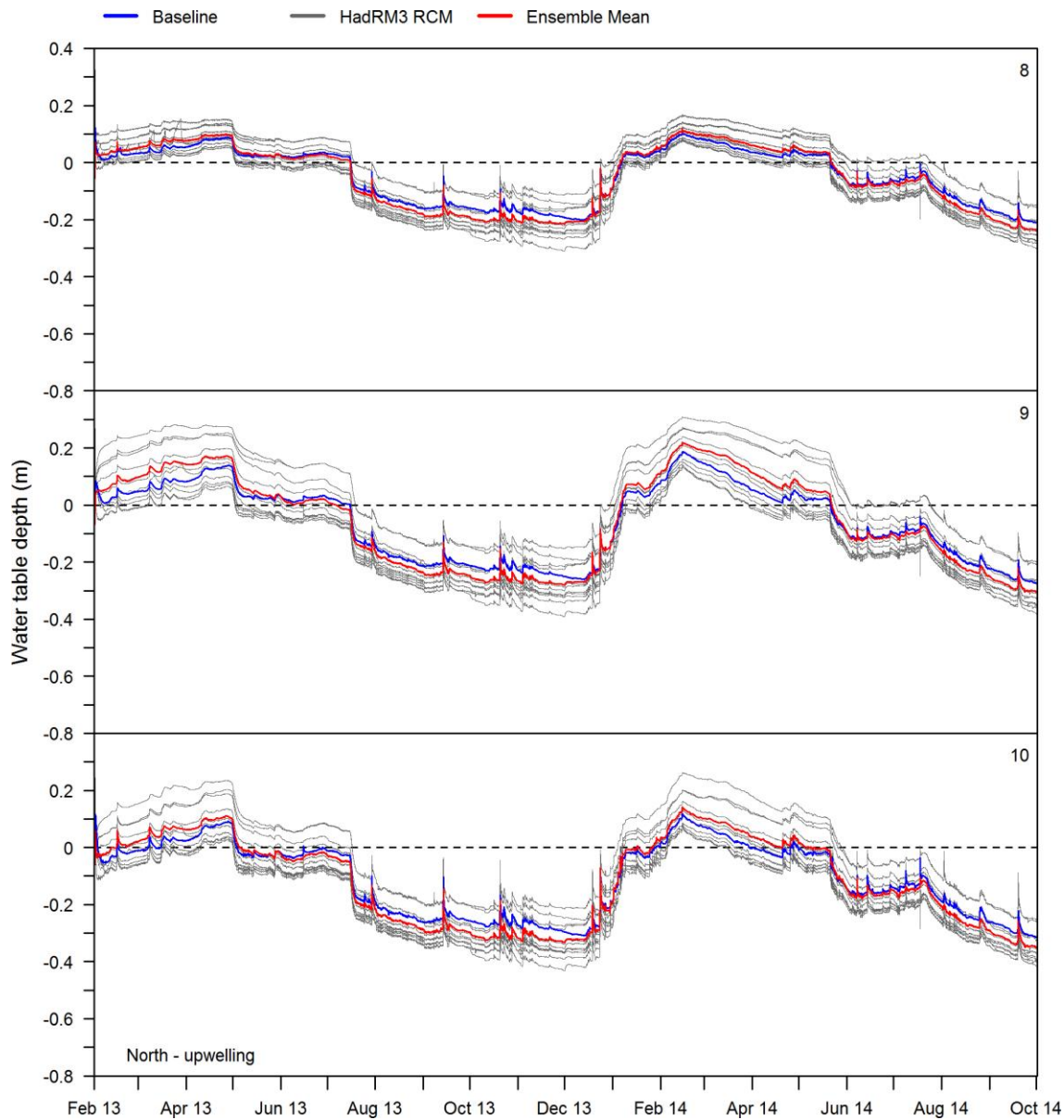


Figure 5-5. Simulated baseline, projected scenario and mean wetland water table depths for North upwelling locations

simulated surface water extends from 1-2 months up to 10 months (locations 1, 2 and 4). Conversely, in these locations, other scenarios (B, G, I, K) project declines in water levels so that they are below ground level for the complete simulation period. The baseline groundwater induced flooding no longer occurs in these locations. In both north and south locations, declines in water level of up to -0.15 m from baseline are simulated during low water level periods in November and December 2013. In the south meadow water levels drop to 0.7 m bgl at location 6 and 0.6 m bgl at locations 5 and 7.

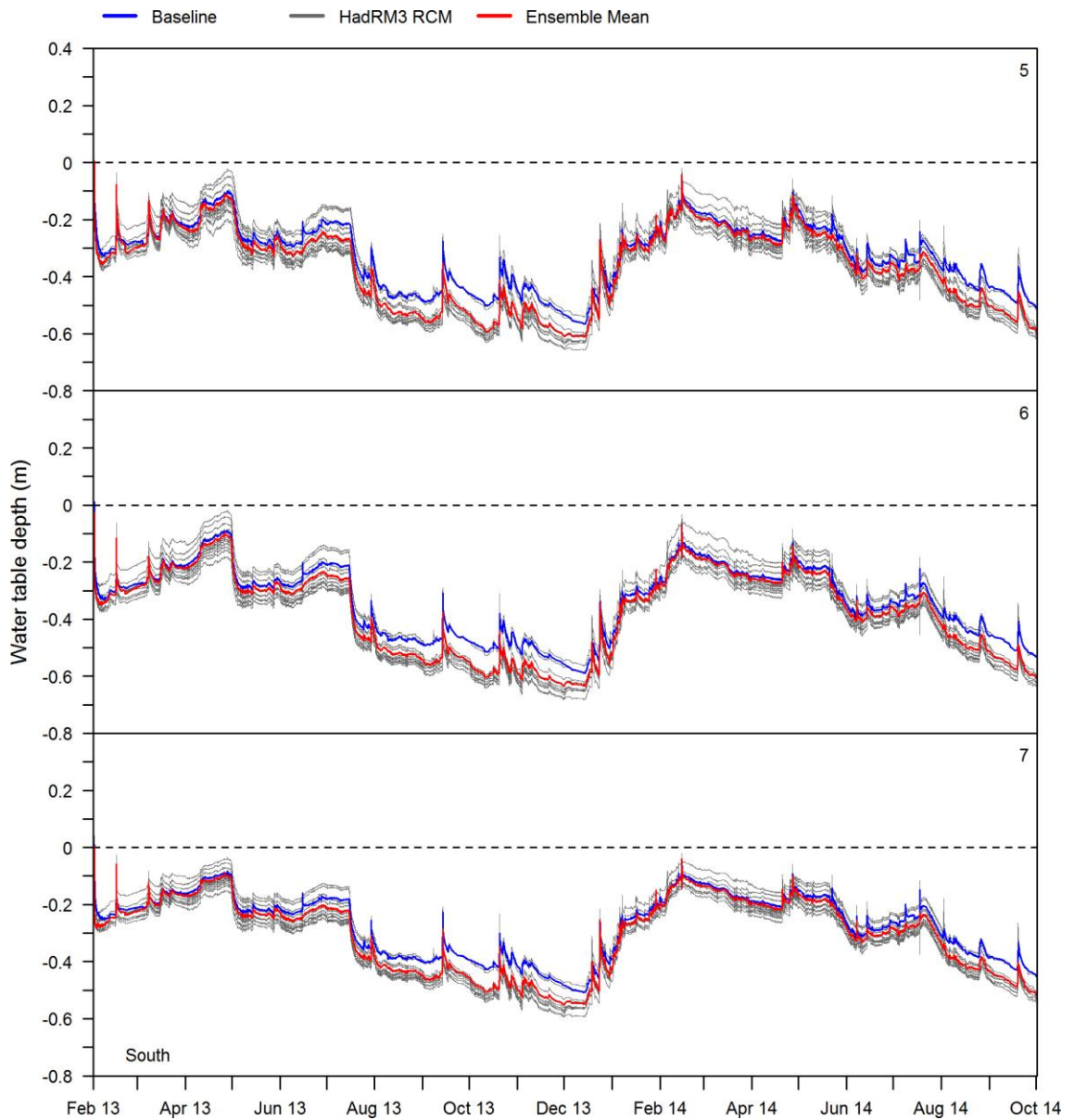


Figure 5-6. Simulated baseline, projected scenario and mean wetland water table depths for South locations

The highest simulated wetland water levels for all locations are generally associated with scenario D. Average increases in levels for this scenario at the different locations for the complete simulation period are in the range +0.03 to +0.15 m. The lowest simulated levels for the north non-upwelling and upwelling areas are associated with scenario K (-0.06 to -0.08 m). In the south meadow, B and I generate the lowest levels over the simulation period (-0.07 m).

Table 5-3. Baseline mean wetland water levels (mBGL) averaged for North no upwelling, North upwelling and South locations, with scenario and mean changes in level (m). *Italicised values indicate negative changes*

Run ID	North - no upwelling	North - upwelling	South
baseline	0.23	0.07	0.31
A	0.04	0.03	-0.03
B	-0.08	-0.06	-0.07
C	0.11	0.08	0.02
D	0.14	0.10	0.04
E	0.07	0.05	-0.01
F	-0.02	-0.02	-0.06
G	-0.07	-0.06	-0.06
H	-0.06	-0.05	-0.06
I	-0.06	-0.05	-0.07
J	-0.01	-0.01	-0.04
K	-0.08	-0.06	-0.06
mean	0	0	-0.03

5.4.2 Channel stage

Simulated stages for baseline and each scenario at seven locations along the Lambourn and three along the Westbrook are shown in Figures 5-7 and 5-8. Mean scenario changes show a general reduction in stage which is much more apparent during periods of low flow (July 2013 – December 2013 and August 2014 – October 2014). For the scenario mean the largest declines in stage occur in mid-December 2013. In the River Lambourn these range from -0.13 (L7) to -0.18 m (L1 and L3), and in the Westbrook from -0.16 (W3) to -0.29 (W2). During periods of high flow the simulated stage for the scenario mean corresponds much more closely to the baseline. Where increases occur they are relatively small, ranging from +0.07 to +0.08 m in all locations except W2 (+0.13 m towards the end of April 2013).

Only two scenarios (C and D) show overall increases through the simulation period (Table 5-4). The largest decreases in stage over the full simulation period in both the Lambourn and the Westbrook are associated with the B and I scenarios (-0.06 and -0.08 m for the Lambourn and Westbrook, respectively). Stage drops to near zero at L1, L3, L4 and W3 in December 2013, the period associated with the largest decline in simulated stage, under scenarios B and I. At the other locations (L2, L5, L6, L7 and W1) the mean magnitude of the declines in stage is from -0.13 for scenario D to -0.2 m for scenario K. During periods of high flow, simulated scenario stages are spread reasonably evenly on either side of the baseline with a maximum range of 0.15 m. At W2 scenario

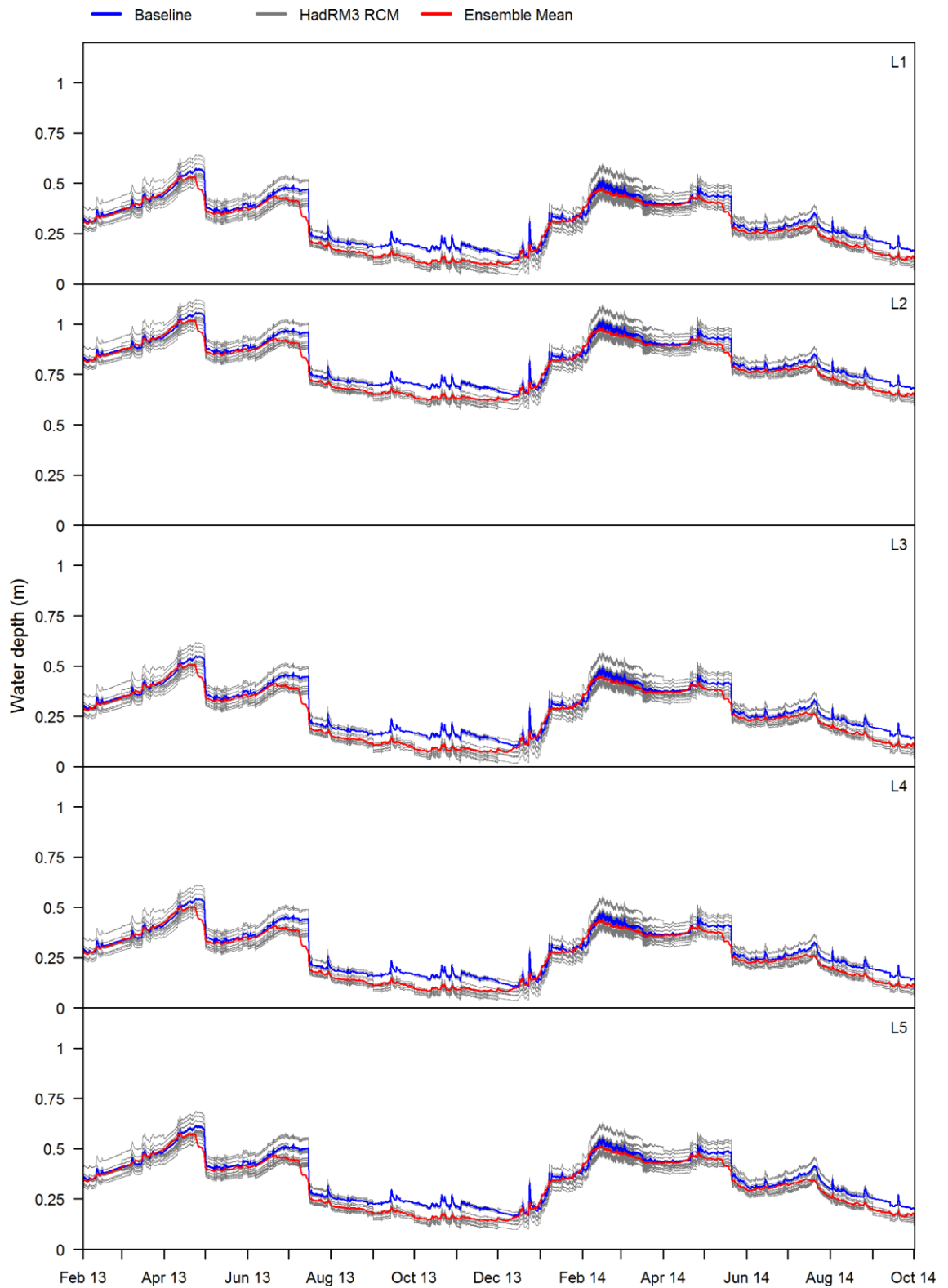


Figure 5-7. Simulated baseline, projected scenario and mean channel stages for River Lambourn locations L1-L5

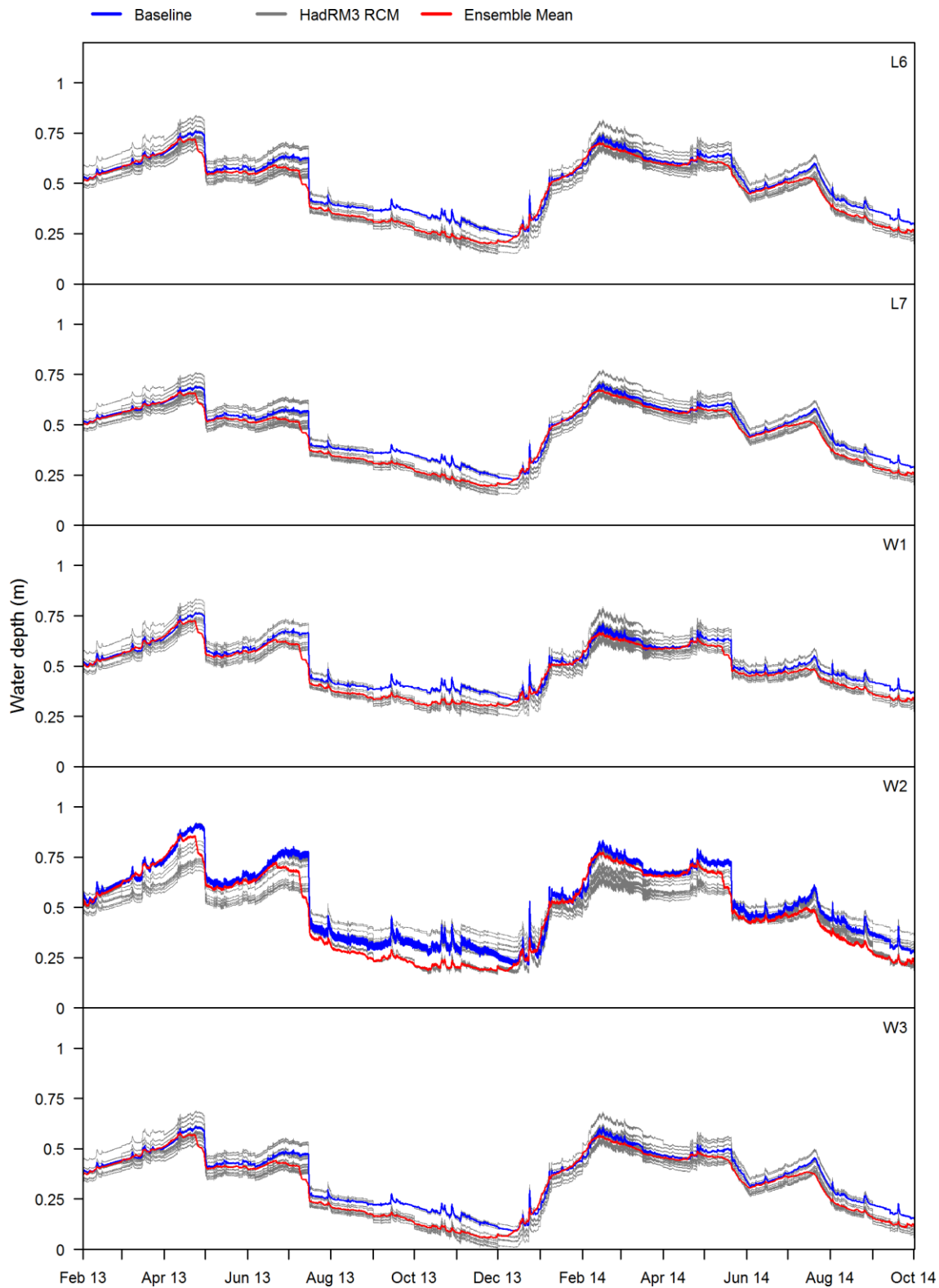


Figure 5-8. Simulated baseline, projected scenario and mean channel stages for River Lambourn locations L6 and L7, and Westbrook locations W1–W3

Table 5-4. Baseline mean channel stage (m) averaged for the River Lambourn and Westbrook, with scenario and mean changes in level (m). Italicised values indicate negative changes

Run ID	Lambourn	Westbrook
baseline	0.44	0.47
A	-0.02	-0.03
B	-0.06	-0.08
C	0.02	0.03
D	0.04	0.05
E	-0.01	-0.01
F	-0.05	-0.06
G	-0.05	-0.06
H	-0.05	-0.07
I	-0.06	-0.08
J	-0.03	-0.04
K	-0.05	-0.06
mean	-0.03	-0.04

levels generally fall below the baseline except for periods of low stage where the A, C, D and E scenarios predict an increase of up to +0.13 m. The December minimum is not as apparent at W2 as with other locations, with the lowest predicted stages occurring through October to December 2013 under the H and I scenarios.

5.4.3 Channel velocity

Changes in velocity for the 11 scenarios display a general decrease through low flow periods, and a greater spread around baseline values through high flow periods (Figures 5-9 and 5-10). Scenario mean projections show greatest decreases, as with stage, in December 2013. In the River Lambourn these range from -0.050 (L7) to -0.035 ms⁻¹ (L3), and in the Westbrook from -0.041 (W2) to -0.039 ms⁻¹ (W1 and W3). At no point does the scenario mean show an increase, although velocities correspond very closely to baseline values at high flow. This is marked in March 2013 at -0.001 ms⁻¹ for L5.

Locations with relatively high initial velocity (L1, L3, L4, L5 and L6) show greater variation in velocity through the simulation period than those with lower initial velocity (L2, L7, W1, W2 and W3). This is especially apparent when comparing the rapid changes in velocity at times of weed cuts which range from a 0.182 ms⁻¹ rise under baseline conditions on 1/5/2013 at L5 to almost no discernible change in the Westbrook locations. Averaged changes show an exaggerated response in the Lambourn compared to the Westbrook (Table 5-6). Increases in velocity for all locations are seen for four scenarios: D and C

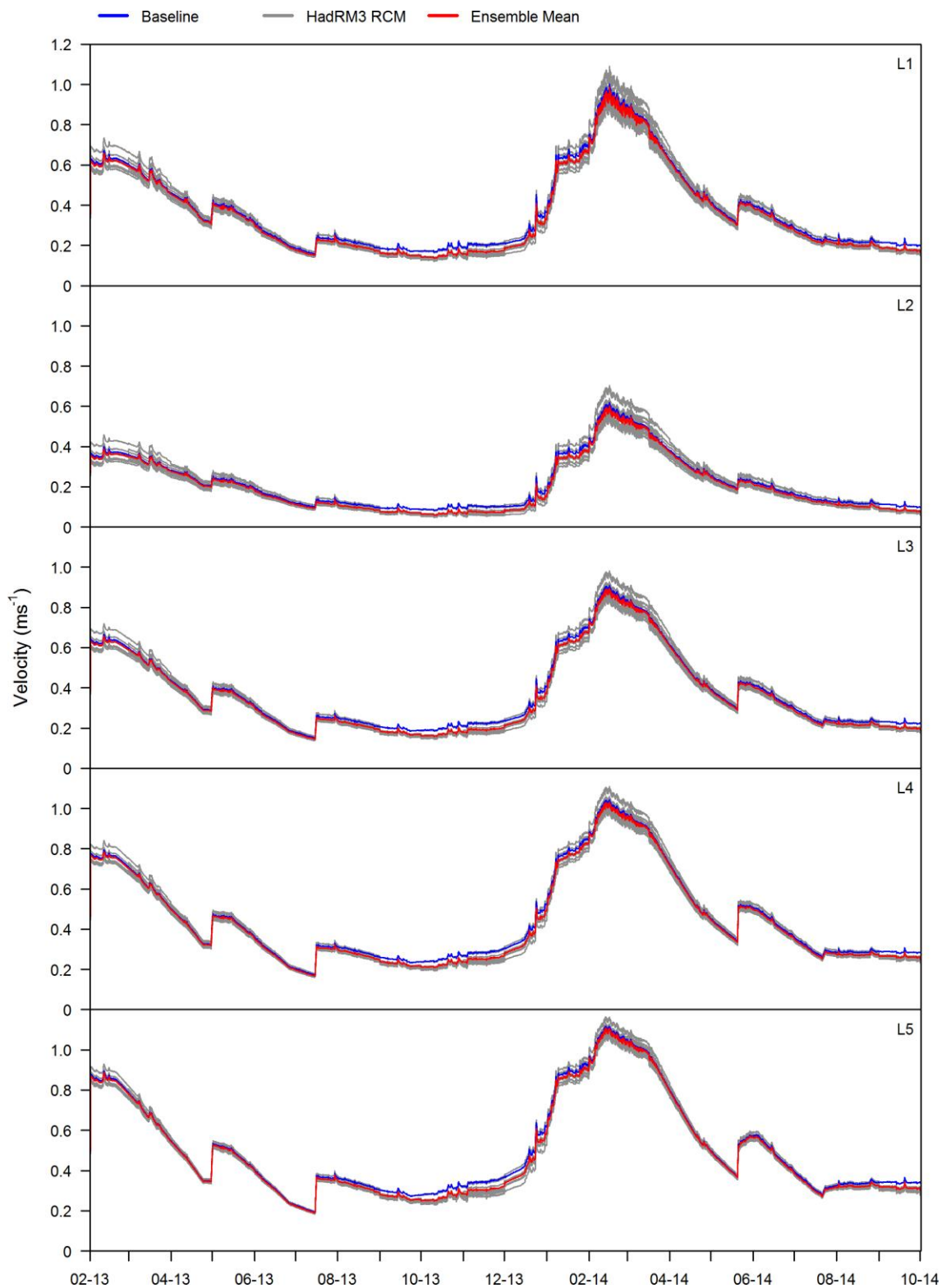


Figure 5-9. Simulated baseline, projected scenario and mean channel velocities for River Lambourn locations L1-L5

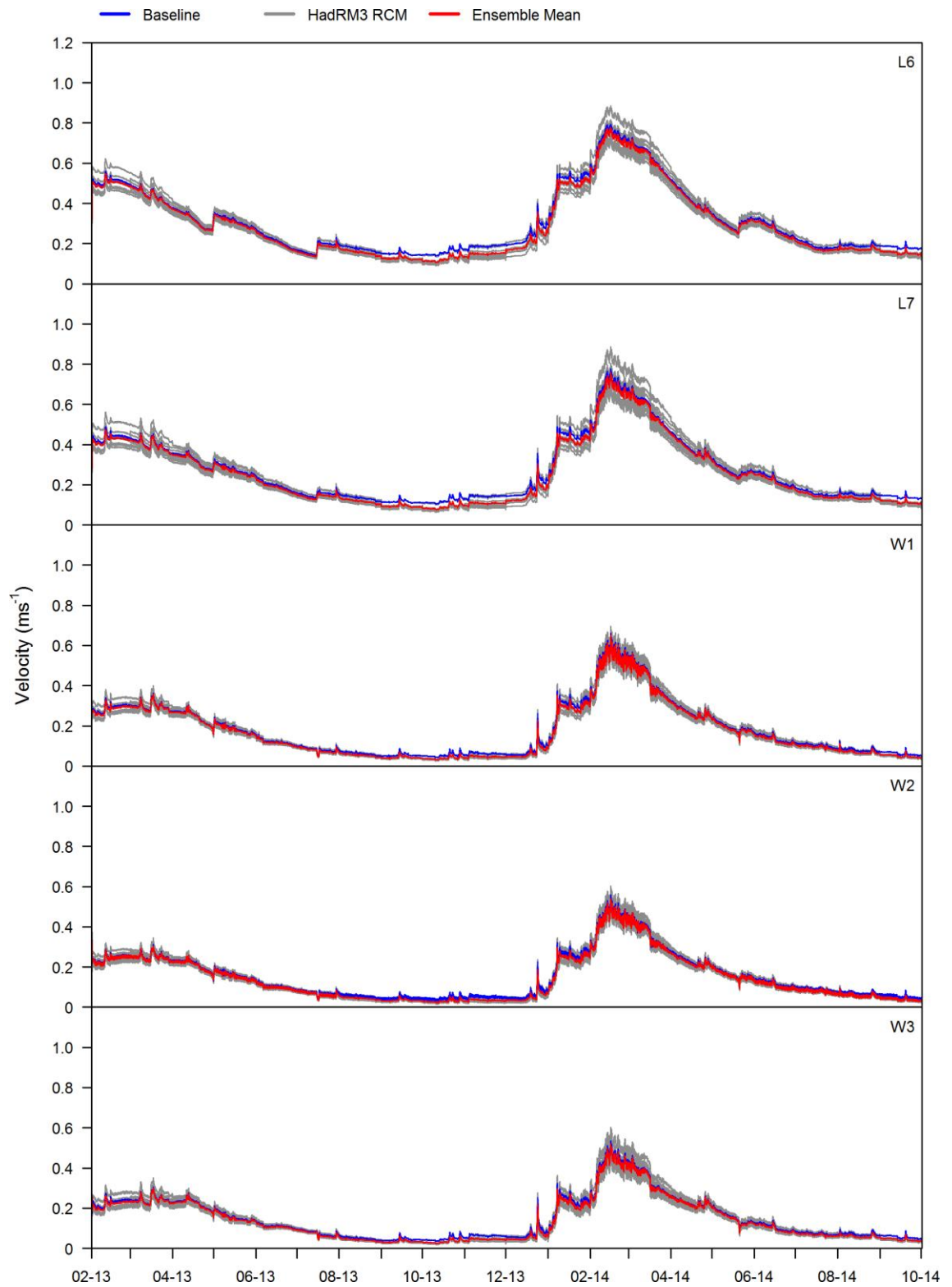


Figure 5-10. Simulated baseline, projected scenario and mean channel stages for River Lambourn locations L6 and L7, and Westbrook locations W1–W3

Table 5-5. Baseline mean channel velocity (ms^{-1}) averaged for the River Lambourn and Westbrook, with individual scenario and mean changes in velocity (ms^{-1}). Italicised values indicate negative changes

Run ID	Lambourn	Westbrook
baseline	0.360	0.159
afgcx	-0.012	-0.006
afixa	-0.033	-0.021
afixc	0.012	0.008
afixh	0.023	0.016
afixi	-0.005	-0.002
afixj	-0.024	-0.014
afixk	-0.022	-0.013
afixl	-0.029	-0.018
afixm	-0.034	-0.022
afixo	-0.016	-0.009
afixq	-0.028	-0.018
mean	-0.014	-0.008

through the whole simulation period, E, March to June, and A, only in April. Of these, the largest increase are associated with scenario D and range from $+0.048$ (L5) to $+0.114 \text{ ms}^{-1}$ (L7) in February 2014. For the other scenarios, the largest decreases in velocity are projected by scenario K, ranging from -0.076 (L2 and L3) to -0.107 ms^{-1} (L7).

5.4.4 Groundwater upwelling

Simulated groundwater flow from the Chalk aquifer in areas where the putty chalk is absent shows a strong seasonality in the baseline, scenario and mean scenarios between winter wet periods and summer dry spells (Figure 5-11). Rapid increases occur during and immediately after weed cuts. A mean scenario increase is evident throughout the simulation period. This is accentuated during periods of high flow (February 2013 – May 2013 and January 2014 – May 2014), with the largest increases in the scenario mean occurring in March 2014.

Of the individual scenarios, three (A, C, D and E) show increases throughout the simulation period. Scenario D displays the largest increases in March 2014 ($+5.41 \text{ mm day}^{-1}$), also the period of greatest inter-scenario variation, with scenario B decreasing by $-0.11 \text{ mm day}^{-1}$. The smallest inter-scenario range occurs in September 2013 during the low flow period (from -0.06 scenario G to $+1.33 \text{ mm day}^{-1}$ scenario C). Only a single scenario (B) shows decreases in groundwater flow over the full simulation

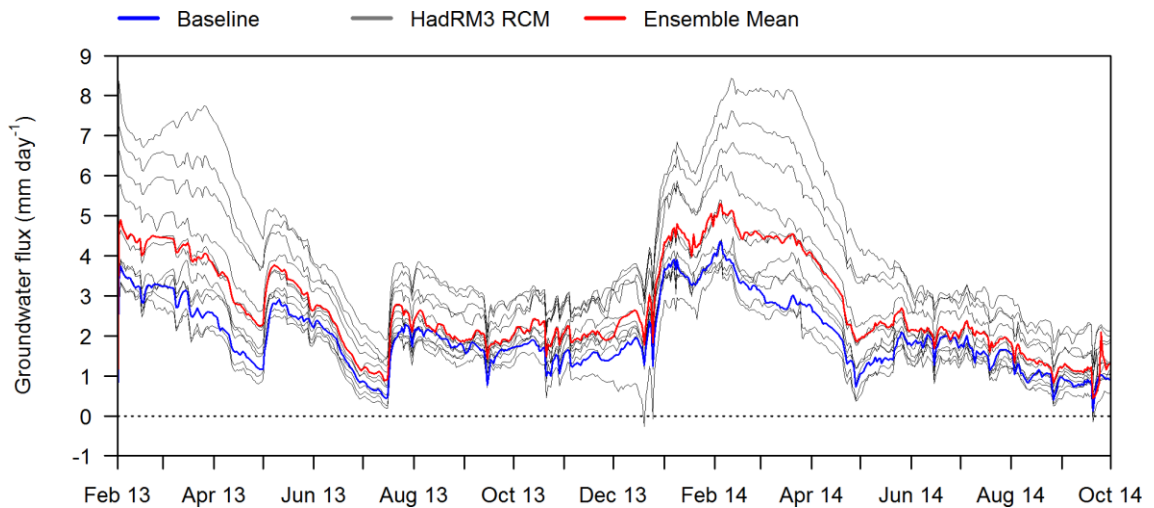


Figure 5-11. Simulated baseline, projected scenario and mean groundwater flow from Chalk aquifer into wetland (positive = upwards)

period, although it is only scenario G that results in negative flow, or recharge, in December 2013.

5.5 Spatial and temporal variations in hydrological response to climate change

Baseline hydrological conditions and in turn the response to climate change differs noticeably over relatively short distances through the wetland. Other studies have shown similar hydrological complexity in comparable settings (Gilvear et al., 1993; Gilvear et al., 1997; Grapes et al., 2006). At these scales hydrological processes are dominated by the interaction between groundwater and surface water, reflecting the site's position in a chalk valley bottom. Indeed, baseline results from the MIKE SHE model indicated proportional contributions to the water balance of 44.2% for surface water, 43.4% for groundwater, 6.3% for precipitation, and 5.7% for actual evapotranspiration (Section 4.5.1). Wetter winters and drier summers due to seasonal changes in scenario precipitation and year-round increasing PET have some direct influence. However, changes to wetland water levels are mostly governed by the projected changes in discharge and groundwater level. These in turn are influenced by meteorological changes occurring over the catchment and regional area. The disadvantage of a hydrological model at the site scale lies in the ability of the boundary conditions to represent flow changes from the wider area. Regional changes in precipitation and evapotranspiration would be expected to translate to comparable changes across the flow boundaries. The comparison of total boundary inflow under each scenario to precipitation shows that changes are within the same magnitude. Additionally, the good

relationship between total boundary inflow, precipitation and evapotranspiration indicates that the effects of climate change are accounted for by the modelling approach.

Inter-scenario member variations in stage are more pronounced than for velocity, suggesting climate induced changes will be depth limited by the hydraulic geometry. Velocities in all scenarios for the simulation period rarely exceed 1 ms^{-1} , indicative of the position of the reach in a lowland catchment with a relatively slow runoff response and large baseflow component. In the south meadow the water levels are principally controlled by boundary channel stages. Scenario changes in water levels in this part of the wetland replicate the pattern of change in the River Lambourn and Westbrook. Conversely, since changes in chalk groundwater levels are larger than those for channel stage, the influence of upwelling chalk groundwater in the north meadow causes a greater projected range of scenario wetland water levels. The relatively small scenario changes in channel stage and velocity indicate the importance of regional and catchment processes in controlling water supply mechanisms for the site. The river has a high base flow index at 0.96 (Marsh and Hannaford, 2008) and an ephemeral source, with the perennial head located 6-7 km downstream. Groundwater feeding the river may act as a buffer to the stresses of climate change at the catchment scale. Therefore at 13 km downstream from the source the effects on discharge would be small. Indeed, a linear regression relationship between changes in the discharge inputs to the MIKE 11 model and the corresponding changes in the groundwater head boundary ($r^2 = 0.77$) is stronger than that for discharge and precipitation minus potential evapotranspiration ($r^2 = 0.63$) (Figure 5-12). Additionally, as discussed in Section 3.5.3, in-channel macrophyte growth is a principal control on river stage at the site, so that the importance of discharge in controlling channel stage and corresponding water levels in the wetland may be moderate.

The uncertainty contained within the projected hydrometeorological drivers for the MIKE SHE model is echoed in the water level responses across the wetland. Channel stage and velocity also reflect some uncertainty in the drivers for discharge inputs to the MIKE 11 model. Inter-scenario variations in simulated water levels differ spatially and over time, exhibiting some seasonality. In the north meadow results are split between four scenarios showing an overall increase in mean water levels throughout the simulation period and the remaining seven that show mean declines. The same directional trend is seen in summer, while in winter six of the 11 members show increases in mean water levels. In the south meadow the general trend of change follows those in the River Lambourn and Westbrook. This is mostly negative with nine out of the 11

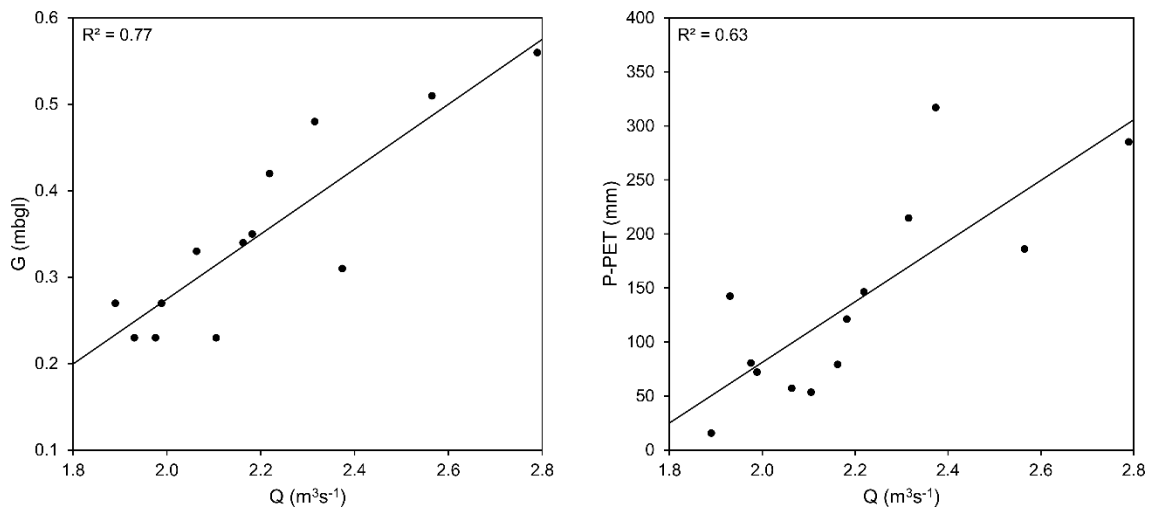


Figure 5-12. Relationships between changes in discharge inputs (Q) and those in the groundwater head boundary (G) and precipitation minus potential evapotranspiration (P - PET)

scenarios resulting in lower mean water levels for the complete simulation period and the summer period. In winter 8 of the 11 scenarios show lower levels, one no appreciable change and only two projecting increases.

Declining river flow and increasing groundwater levels as indicated by the hydrometeorological projections are counterintuitive. The high reported baseflow of the River Lambourn would, at first glance, lead the reverse relationship to be expected. However, the proportion of this baseflow which comes from the gravel aquifer or the Chalk aquifer is unclear, as the hydrochemistry in the gravels is well mixed and displays similarity to the Chalk aquifer (Section 3.8.2). The gravel aquifer itself accounts for a down-valley component of groundwater flow, with variable hydraulic connection to the Chalk (Grapes et al., 2006), whilst the river is in good hydraulic connectivity with the gravels (Allen et al., 2010). It is possible that the two aquifers will experience differing responses to climate change, with the effect shown that the mostly gravel aquifer influenced river will display reductions in discharge, whilst the mostly separated Chalk aquifer will show increases in head.

5.6 Summary

Projected changes in precipitation, potential evapotranspiration, channel discharge and groundwater level were derived from the UK Climate Projections 2009 ensemble of climate models for the 2080s under different scenarios. These were applied as inputs to the MIKE SHE distributed hydrological/hydraulic model of the Lambourn Observatory to investigate how climate change scenarios affect hydrological functioning. The simulated hydrological impacts of climate change vary considerably over relatively small distances

within the Observatory. This is due to differences in groundwater/surface water interaction and water availability, and reflect the site's position in a chalk valley bottom. Changes to the hydrological functioning of the Observatory are influenced by meteorological changes occurring over the catchment and regional area. It is shown that the modelling approach accounts for the regional effects of climate change at the site scale.

Discrete areas of groundwater upwelling in the North are associated with an exaggerated response of water levels to climate change compared to non-upwelling areas. These are coincident with regions where the weathered chalk layer, which otherwise separates two main aquifers, is absent. Scenario changes in water levels in the South of the wetland replicate the pattern of change in the River Lambourn and Westbrook. Relatively small scenario changes in channel stage again indicate the importance of wider scale processes in controlling water supply mechanisms for the Observatory. Declining river flow and increasing groundwater levels as indicated by the hydrometeorological projections may be due to differences in hydraulic connectivity between the Chalk aquifer, gravel aquifer and river. The two aquifers may well respond differently to climate change, with the effect shown that the mostly gravel aquifer influenced river will display small reductions in discharge, whilst the mostly separated Chalk aquifer will show relatively large increases in head.

The different hydrological impacts of climate change in distinct areas of such a relatively small site will have important implications for the maintenance of conservation priority and productive species and communities. Differences in water level requirements between communities and species implies diverse ecological responses to climate change, a topic which forms the focus of the following chapter.

Chapter 6

Ecological impacts of climate change

6.1 Introduction

As discussed in Section 1.4.1, wetlands are highly vulnerable to climate change due to the primary importance of the hydrological regime in controlling their ecological characteristics (e.g. Baker et al., 2009), while climate change is likely to impact fluvial ecosystems through changes in the flow regime. Hydrological changes due to climate change may be linked to water level requirements of different wetland species and communities to infer ecological impacts (Acreman et al., 2009; Wheeler et al., 2004). For instance, water table level regime is a dominant control on wetland plant communities (Silvertown et al., 1999) (Section 1.3.2.1). Whilst hydrological modelling has been used to assess some ecological impacts of climate change, in many cases this has not been undertaken at a resolution sufficient to directly infer impacts for particular species and communities; instead surmising effects through changes in habitat availability (Barron et al., 2012; Candela et al., 2009; Johnson et al., 2005). Other studies have postulated impacts generalised over regional scales (Acreman et al., 2009; Herrera-Pantoja et al., 2012).

In watercourses, a direct relationship between physical habitat and flow enables assessments of the ecological responses to changes in the flow regime (Beecher et al., 1993; Cavendish and Duncan, 1986). Hydraulic changes due to climate change may be linked to the depth and velocity requirements for different species and provide a measure of available physical habitat as a function of flow. The hydraulic components of the MIKE SHE hydrological model of the Observatory may be used to assess the impacts of climate change on physical habitat using standard outputs of flow, depth and velocity. A physical habitat-discharge relationship may thus be produced that can incorporate dynamic processes modelled within a river, such as macrophyte growth. The idea of using existing hydraulic models for physical habitat assessment has existed for 20 years (Dunbar et al., 1997), yet, to the author's knowledge, never been operationalised. This further meets a priority within the Natura 2000 Site Improvement Assessment Plan where investigating the impact of climate change on ecology (via hydrological changes) is stated as an agreed measure (Natural England, 2014).

In this chapter simulated water levels under each of the climate change scenario investigated in Chapter 5 are compared to the requirements of conservation

species / communities for which the Observatory wetland is designated. In addition, the effects of climate change on physical habitat for brown trout (*Salmo trutta*) are assessed for the reach of the River Lambourn contained within the Observatory. An assessment is provided of the potential ecohydrological effects of climate change upon the wetland and river.

6.2 Assessment of wetland ecological impacts of climate change

6.2.1 Water level requirements for wetland species and communities

Simulated peat water table levels for both the baseline and each climate change scenario were compared to the water level requirements for the MG8 community which, as described in Section 2.1, contributes to the site's scientific and nature conservation status. These water level requirements are defined as monthly water table depth zones (Wheeler et al., 2004; Wheeler et al., 2009). Figure 6-1 shows the requirements for the MG8 community as defined by Wheeler et al. (2004), with green areas indicating desirable conditions, amber representing tolerable conditions should the water table fall within these zones for limited periods, and red indicative of intolerable conditions. Analysis centred on establishing the favourability of current (baseline) water table conditions to supporting this community and whether climate change-related modifications to water tables are likely to cause a shift in hydrological conditions which could have implications for MG8 species.

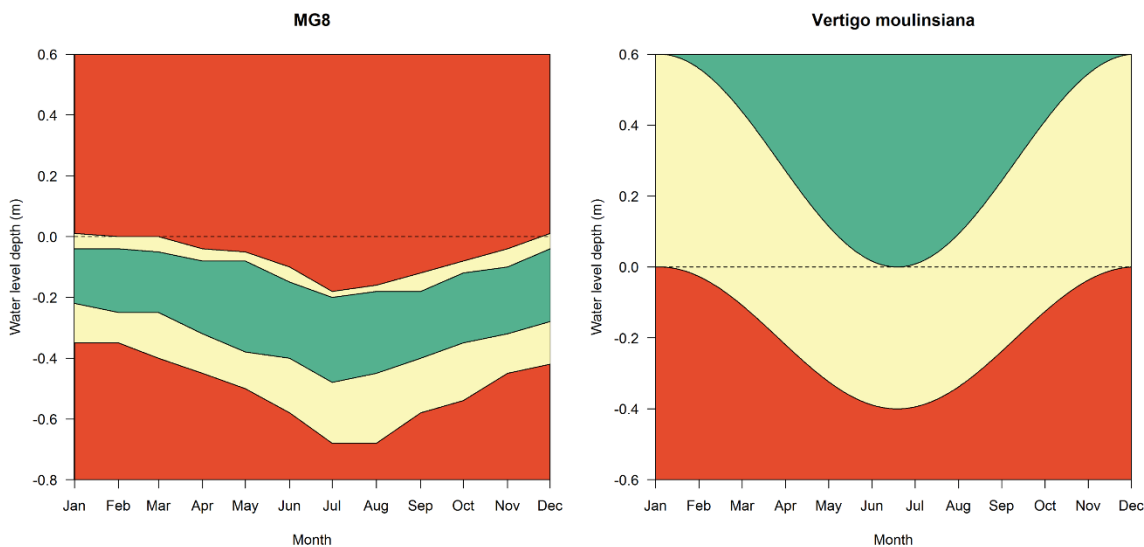


Figure 6-1. Water level requirements for the MG8 vegetation community (after Wheeler et al., 2004) and Desmoulin's whorl snail (after Tattersfield and McInnes, 2003). Red - intolerable; Amber - tolerable for limited periods; Green - desirable

Simulated peat water levels were also compared to the hydrological requirements of the conservation relevant Desmoulin's whorl snail (*Vertigo moulinsiana*). Tattersfield and McInnes (2003) suggested that optimal conditions for the snail occur where water levels are continuously above ground level, fluctuating between 0.6 and 0.0 m in winter and summer respectively. Suboptimal, yet tolerable, conditions exist where water levels fluctuate between 0.2 m above ground in winter and 0.2 m below ground in summer. If water levels drop below ground level in winter and are more than 0.4 m below the surface in summer the snail is unlikely to be present. These suggested conditions were used to define monthly ranges of desirable, tolerable and intolerable water levels for the Desmoulin's whorl snail in the same form as those used for the MG8 vegetation community (Figure 6-1ii). This enabled the same approach for defining the suitability or otherwise of baseline and scenario water level regimes for this individual species.

6.2.2 Climate change impacts on vegetation community

Inspection of simulated wetland water levels against a backdrop of water depth zones for the MG8 vegetation community reveals that ecological responses fall into three spatial groups, following the water level responses (Section 5.4.1): locations in the north meadow characterised by upwelling groundwater (North – Upwelling; Figure 6-2), locations in the north meadow where upwelling is absent (North – no upwelling; Figure 6-3), and locations in the south of the wetland (South; Figure 6-4). Under baseline conditions water levels in the North – no upwelling and South locations are, on the whole, within the desirable or tolerable ranges for MG8 vegetation (Figures 6-3 and 6-4). They are, however, often close to the boundary of the intolerable zone suggesting that current conditions are approaching the limit for this community. Water levels in the South fall into the lower intolerable zone for 6.6% of the simulated period. This occurs in December 2013 and coincides with the lowest simulated water levels (Table 6-1). In the North – no upwelling locations simulated baseline water levels extend into the higher intolerable zone during peak periods in April 2013, June 2013 and February 2014 (Figure 6-2). These periods account for up to 16.0% of the total simulation period. At other locations groundwater upwelling elevates water levels so that they are above the tolerable range for much of the period, dropping to tolerable conditions for between 6 and 12 months during the summer low periods.

The scenario ranges show that the potential effects of climate change on MG8 vegetation differ across the site. However, in nearly all scenarios there is a shift towards more prolonged intolerable conditions for this particular vegetation community. In the locations that experience groundwater upwelling the higher groundwater levels within the

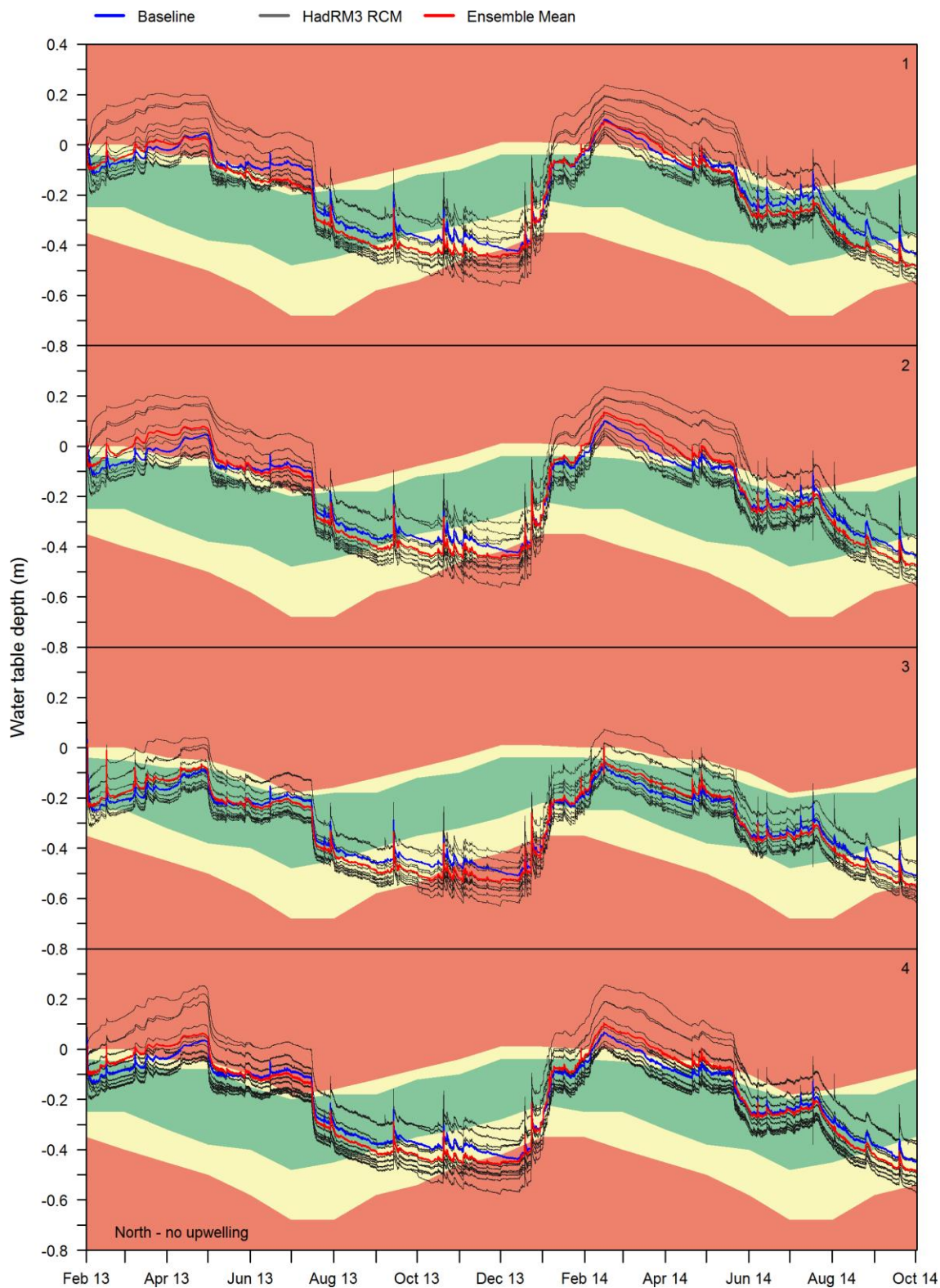


Figure 6-2. Simulated baseline, projected scenario and mean wetland water table depths for North no upwelling locations superimposed over the MG8 vegetation community water level requirements zone diagrams. Red - intolerable; Amber - tolerable for limited periods; Green - desirable

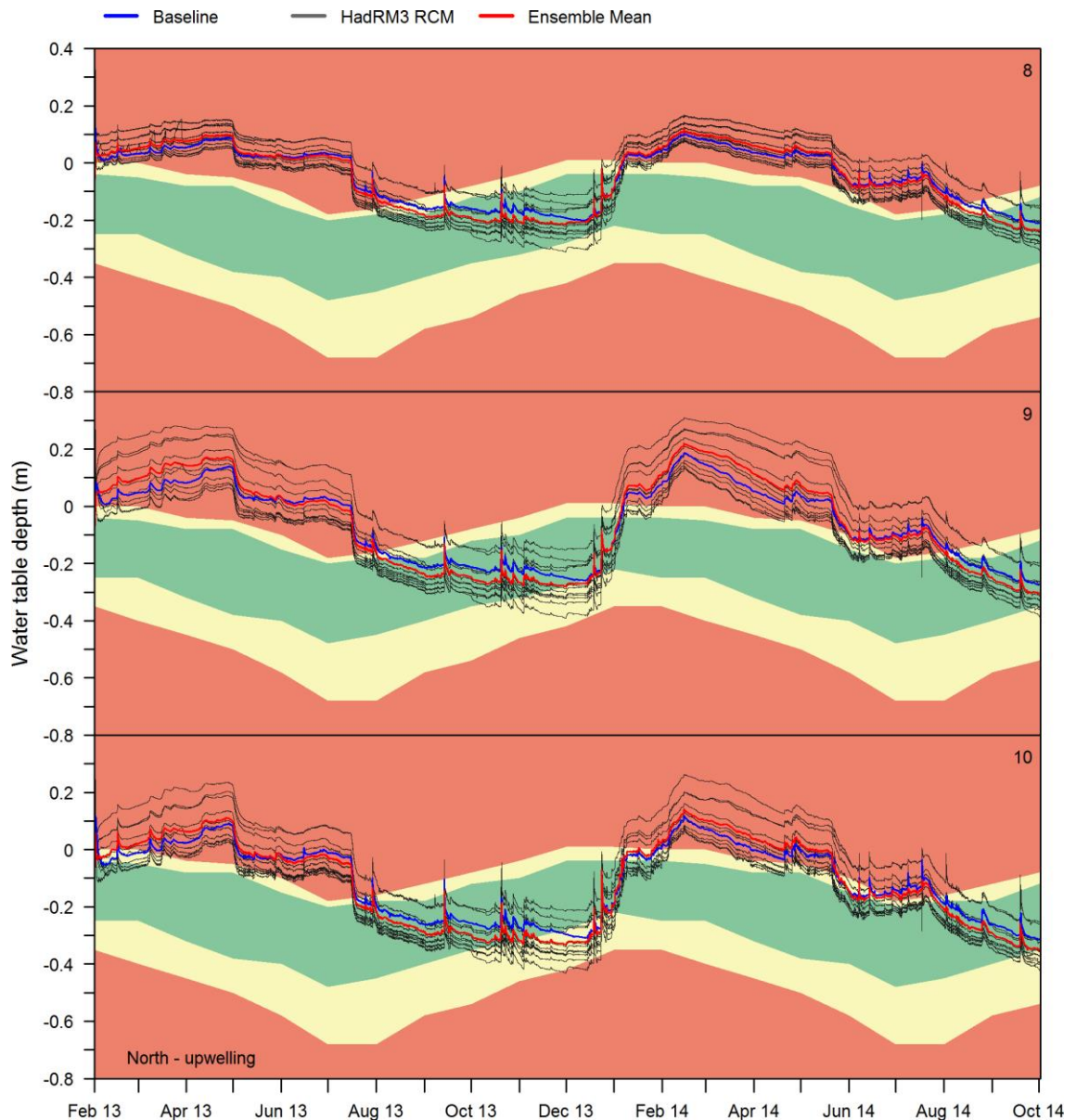


Figure 6-3. Simulated baseline, projected scenario and mean wetland water table depths for North upwelling locations superimposed over the MG8 vegetation community water level requirements zone diagrams. Red - intolerable; Amber - tolerable for limited periods; Green – desirable

underlying chalk for some scenarios push the highest wetland water levels further out of tolerable limits. The durations of the periods when water levels are in the upper intolerable zone therefore increases for scenarios A, C, D and E. However, for most scenarios the lower levels at other times of year now extend into the tolerable conditions for a larger proportion of the simulated period. In North - no upwelling locations the upper range of changes increases both the magnitude and duration of water levels falling within the intolerable zone. For the C, D and E scenarios water levels are within

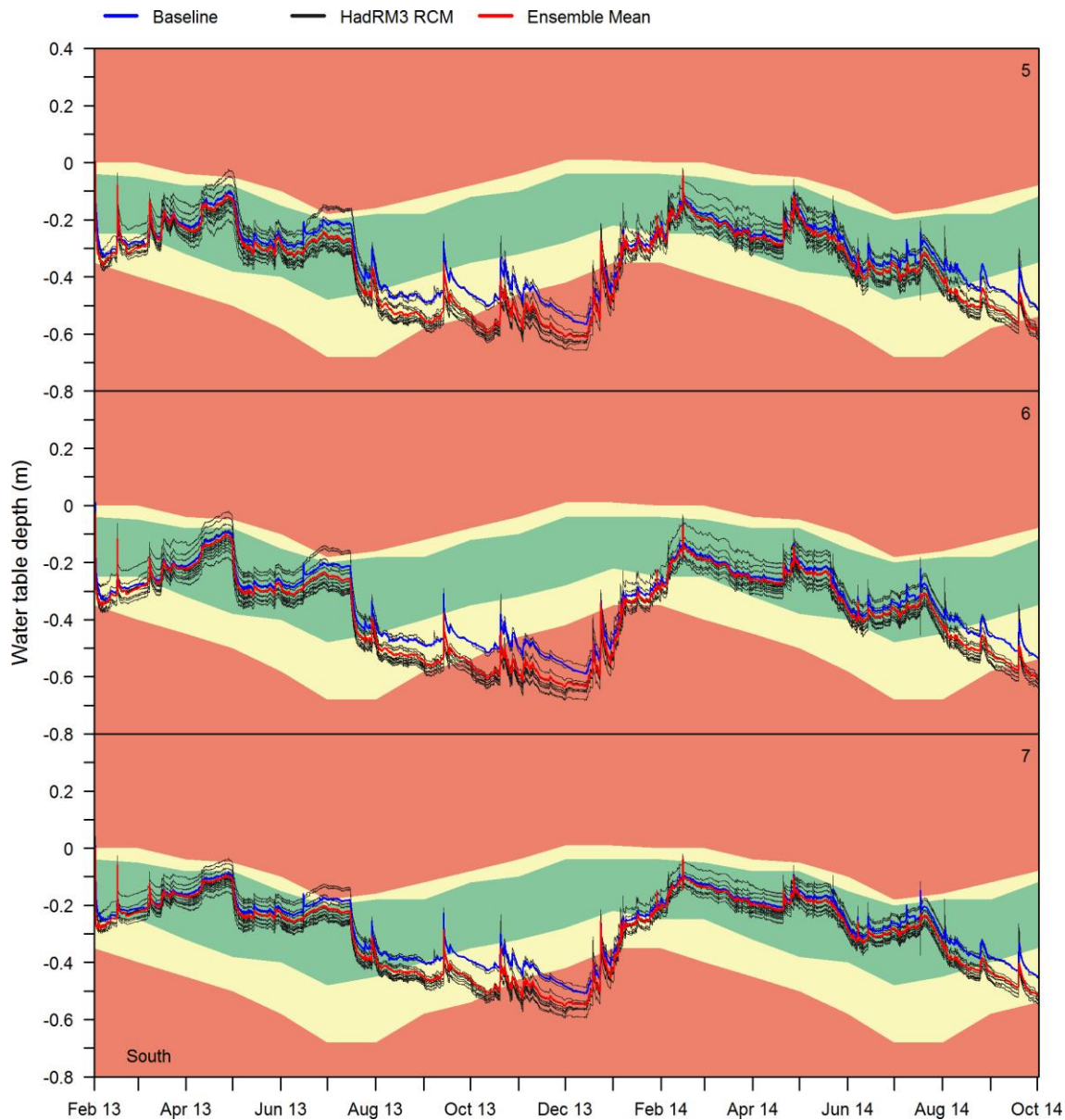


Figure 6-4. Simulated baseline, projected scenario and mean wetland water table depths for South locations superimposed over the MG8 vegetation community water level requirements zone diagrams. Red - intolerable; Amber - tolerable for limited periods; Green – desirable

this zone for as much as 8-10 months. At the other extreme, the lower levels associated with some scenarios increase the occurrence of tolerable rather than desirable conditions, and pushes levels into the intolerable zone through the October – December 2013 low period. In the south meadow the projected increases in water levels for scenarios C and D could be beneficial for the MG8 vegetation community since the water table moves into the desirable zone. However, all of the other scenarios predict a decrease from desirable to tolerable levels, with a longer duration inside the tolerable zone of up to 4 months.

Table 6-1. Percentage of full simulation period (01 Feb 2013 – 01 Oct 2014) simulated baseline, scenario and mean water levels are within each water depth zone (WDZ) for the MG8 plant community at all piezometer locations. UI, Upper Intolerable; UT, Upper Tolerable; D, Desirable; LT, Lower Tolerable; LI, Lower Intolerable

Run ID	WDZ	North - no upwelling				North - upwelling			South		
		1	2	3	4	8	9	10	5	6	7
baseline	UI	5.3	16.0	0.1	12.6	71.3	60.7	39.9	0.0	0.0	0.0
	UT	9.2	13.7	0.0	9.5	10.8	6.2	17.6	0.0	0.1	0.1
	D	69.3	57.4	72.4	62.5	17.9	33.1	41.9	62.6	58.8	80.5
	LT	16.1	12.9	22.5	15.4	0.0	0.0	0.6	30.8	32.8	15.3
A	LI	0.0	0.0	5.0	0.0	0.0	0.0	0.0	6.6	8.2	4.1
	UI	22.2	43.5	0.1	31.0	74.4	65.2	53.5	0.0	0.0	0.0
	UT	23.2	6.7	3.3	16.7	6.7	5.8	6.8	0.0	0.1	0.1
	D	40.4	35.7	70.5	36.5	18.9	29.0	39.6	55.2	53.6	70.3
B	LT	14.3	14.0	21.0	15.8	0.0	0.0	0.0	33.8	31.7	22.4
	LI	0.0	0.0	5.1	0.0	0.0	0.0	0.0	11.0	14.6	7.2
	UI	0.1	2.4	0.1	0.2	48.3	31.2	14.1	0.0	0.0	0.1
	UT	1.8	6.4	0.0	6.7	12.8	20.5	12.4	0.0	0.1	0.0
C	D	69.2	64.2	47.4	64.3	38.8	45.2	65.0	37.1	35.1	62.4
	LT	23.5	22.1	40.0	22.8	0.0	3.1	8.5	48.0	46.8	29.7
	LI	5.4	4.9	12.4	6.0	0.0	0.0	0.0	14.9	18.0	7.8
	UI	51.5	55.4	5.4	49.2	82.7	78.4	68.1	0.0	0.0	0.0
D	UT	7.3	6.6	12.4	10.2	8.1	6.1	5.5	2.3	2.8	5.2
	D	35.3	32.5	64.0	34.0	9.1	15.5	26.4	61.5	59.6	75.8
	LT	6.0	5.5	16.5	6.5	0.0	0.0	0.0	30.3	29.2	14.8
	LI	0.0	0.0	1.7	0.0	0.0	0.0	0.0	5.9	8.3	4.2
E	UI	52.5	56.6	17.9	54.0	84.5	78.9	68.8	0.7	1.1	0.1
	UT	6.7	5.9	13.2	6.2	9.9	8.4	5.6	3.4	4.7	6.9
	D	40.8	37.4	51.7	37.9	5.7	12.8	25.5	68.4	62.1	75.9
	LT	0.0	0.1	17.2	1.8	0.0	0.0	0.0	22.8	24.7	14.0
F	LI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.6	7.3	3.2
	UI	46.5	50.1	4.3	44.6	74.9	69.6	58.9	0.0	0.0	0.0
	UT	4.3	3.0	10.0	6.4	7.2	3.0	5.6	0.2	1.1	0.9
	D	35.8	34.0	60.1	33.3	17.9	27.3	35.5	57.5	55.7	70.7
G	LT	13.4	12.9	21.2	15.6	0.0	0.0	0.0	33.1	29.7	21.9
	LI	0.0	0.0	4.4	0.0	0.0	0.0	0.0	9.1	13.5	6.5
	UI	10.6	18.6	0.0	13.6	60.7	51.8	40.2	0.0	0.0	0.0
	UT	8.8	13.6	0.1	9.1	7.2	7.2	10.3	0.0	0.0	0.0
H	D	56.9	45.6	63.3	52.6	32.0	41.0	42.9	41.9	40.7	66.5
	LT	22.8	21.1	27.5	21.1	0.0	0.0	6.6	43.7	42.7	25.6
	LI	0.8	1.2	9.1	3.7	0.0	0.0	0.0	14.4	16.6	7.9
	UI	6.5	16.1	0.1	10.8	52.4	46.1	29.5	0.0	0.0	0.0
I	UT	10.1	5.9	0.0	6.9	10.0	6.7	14.4	0.0	0.1	0.1
	D	53.9	51.2	54.2	54.1	37.6	38.5	42.5	41.6	41.8	68.8
	LT	20.5	18.8	30.2	18.6	0.0	8.8	13.6	44.9	41.9	22.4
	LI	9.0	8.1	15.4	9.5	0.0	0.0	0.0	13.5	16.2	8.7
J	UI	1.8	9.4	0.1	5.9	57.0	48.4	22.0	0.0	0.0	0.0
	UT	6.3	8.0	0.0	5.1	7.8	6.3	21.5	0.0	0.0	0.1
	D	66.0	57.7	56.0	63.3	35.2	44.1	48.4	39.6	36.9	66.3
	LT	21.0	20.2	31.8	20.2	0.0	1.1	8.1	45.9	45.3	25.9
K	LI	5.0	4.7	12.0	5.6	0.0	0.0	0.0	14.4	17.8	7.8
	UI	0.7	5.7	0.1	1.9	54.5	43.0	16.9	0.0	0.0	0.0
	UT	3.9	6.7	0.0	6.7	10.6	12.0	19.4	0.0	0.0	0.1
	D	71.1	65.5	52.9	66.3	34.9	44.6	56.4	35.6	32.3	64.4
L	LT	19.7	17.7	36.9	19.7	0.0	0.4	7.2	49.7	50.1	27.5
	LI	4.6	4.3	10.1	5.4	0.0	0.0	0.0	14.6	17.6	8.0
	UI	16.2	21.4	0.0	16.2	64.3	55.9	44.5	0.0	0.0	0.0
	UT	6.6	17.0	0.1	14.8	7.8	5.6	8.6	0.0	0.0	0.0
M	D	56.5	42.0	67.1	47.8	27.9	38.5	42.1	50.2	47.7	69.6
	LT	20.5	19.4	25.0	19.2	0.0	0.0	4.7	37.7	36.9	23.5
	LI	0.3	0.1	7.8	2.1	0.0	0.0	0.0	12.1	15.4	6.9
	UI	0.1	4.0	0.1	1.6	49.9	36.5	15.8	0.0	0.0	0.1
N	UT	2.3	6.7	0.0	6.2	13.4	16.0	17.8	0.0	0.1	0.0
	D	69.4	63.6	51.3	65.5	36.7	41.7	56.5	42.8	41.0	65.7
	LT	21.8	19.4	36.4	20.1	0.0	5.9	9.9	42.1	41.2	24.9
	LI	6.4	6.3	12.2	6.7	0.0	0.0	0.0	15.1	17.7	9.3
mean	UI	16.3	25.9	0.0	16.9	67.9	60.5	46.6	0.0	0.0	0.0
	UT	8.0	15.6	0.6	17.5	7.2	6.1	9.7	0.0	0.0	0.0
	D	53.6	38.6	65.6	43.1	24.9	33.4	39.8	52.4	49.8	65.4
	LT	21.8	19.9	24.7	21.8	0.0	0.0	3.9	34.0	32.4	26.1
	LI	0.3	0.0	9.2	0.7	0.0	0.0	0.0	13.6	17.7	8.5

6.2.3 Climate change impacts on Desmoulin's whorl snail

Examination of simulated baseline and scenario water levels against the water level requirements of Desmoulin's whorl snail shows that in North – upwelling locations simulated levels indicate baseline conditions are, on the whole, tolerable for the snail (Figure 6-5). Levels only dip into the intolerable zone between November 2013 and January 2014, accounting for between 12.9% and 25% of the simulated period (Table 6-2). The largest increases in level under climate change suggest improved conditions for the snail with levels just reaching the desirable zone for short periods in June to August 2013. For scenario D water levels are within the desirable zone for 9.9% of the simulation period. Conversely the largest decreases in level from the individual scenarios causes an earlier departure (from October 2013 instead of November 2013) into the intolerable zone and suggest the shift to intolerable conditions (albeit by small amounts) at the beginning and end of the simulation period.

For North – no upwelling locations both baseline and scenario simulated water levels are, for most of the simulation period, within the tolerable zone, while in the South they predominantly fall into the intolerable zone (Figures 6-6 and 6-7). The increases in water levels for 4 out of the 11 climate change scenarios (A, C, D and E) simulated for North – no upwelling locations have the potential to improve conditions for the snail, with predicted increases in the duration of tolerable conditions ranging between +5.0% (A) and +11.6% (C) (Table 6-2). Only two scenarios (C and D) show water level increases into the desirable zone. Where scenarios display lower water levels through the year the duration of tolerable conditions decreases, especially for scenario B where the duration of the period when water levels are within the intolerable zone increases by +17.4%.

In South locations, where baseline and scenario water levels do not intercept the ground surface, conditions approach tolerable on few occasions. For the baseline these are at the high points between April and June 2013 and again between May and August 2014. The largest increases in climate change scenario water levels do little to improve conditions for the snail, only slightly extending the duration of tolerable conditions for the scenario with the largest increases (D) by +8.7%. Scenarios with the largest declines in water levels (B, I) cause water levels to extend further into the lower intolerable zone, increasing duration by up to 9.8% (3.25 months).

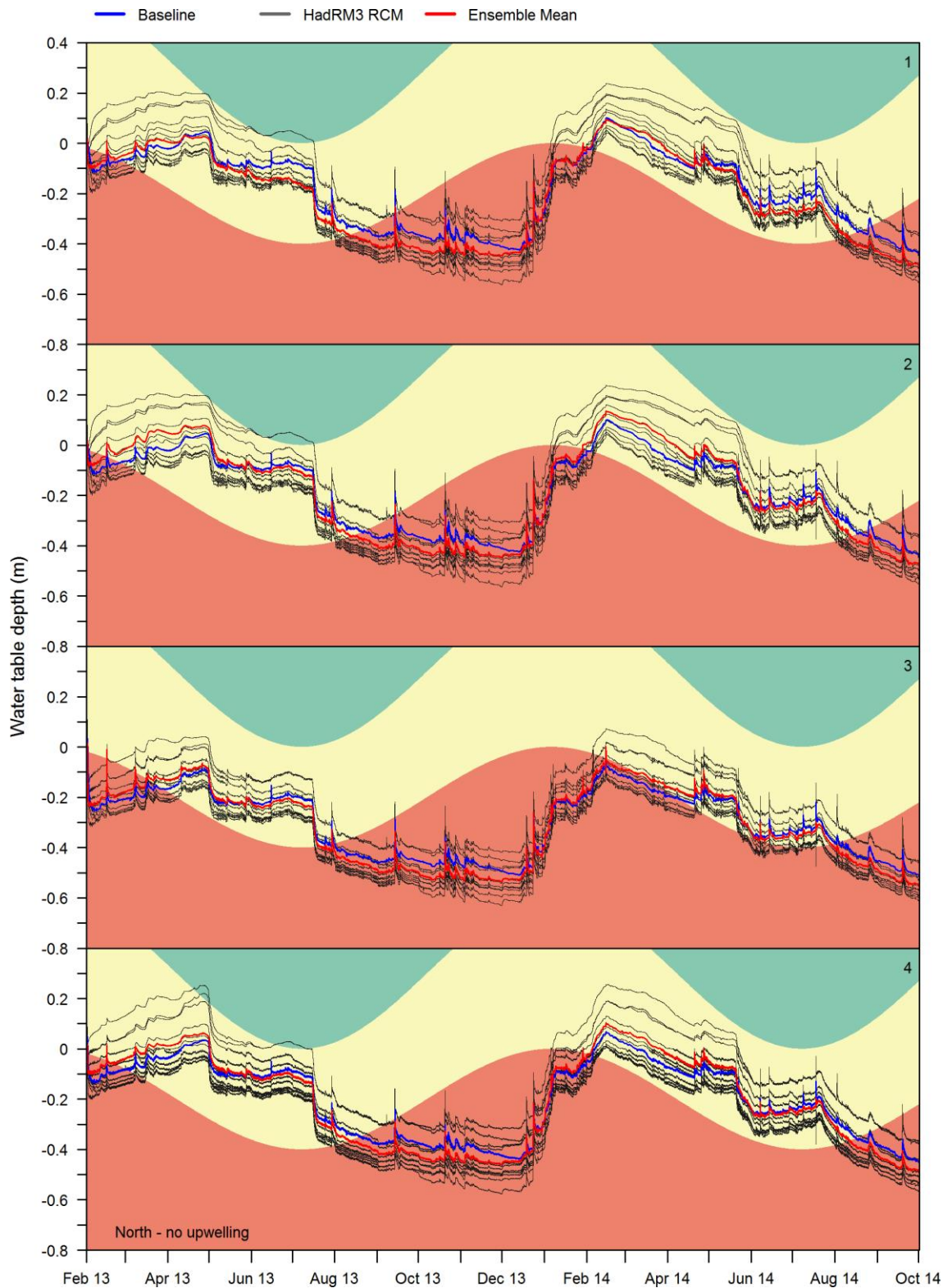


Figure 6-5. Simulated baseline, projected scenario and mean wetland water table depths for North no upwelling locations superimposed over the Desmoulin's whorl snail (*Vertigo moulinsiana*) water level requirements zone diagrams. Red - intolerable; Amber - tolerable for limited periods; Green - desirable

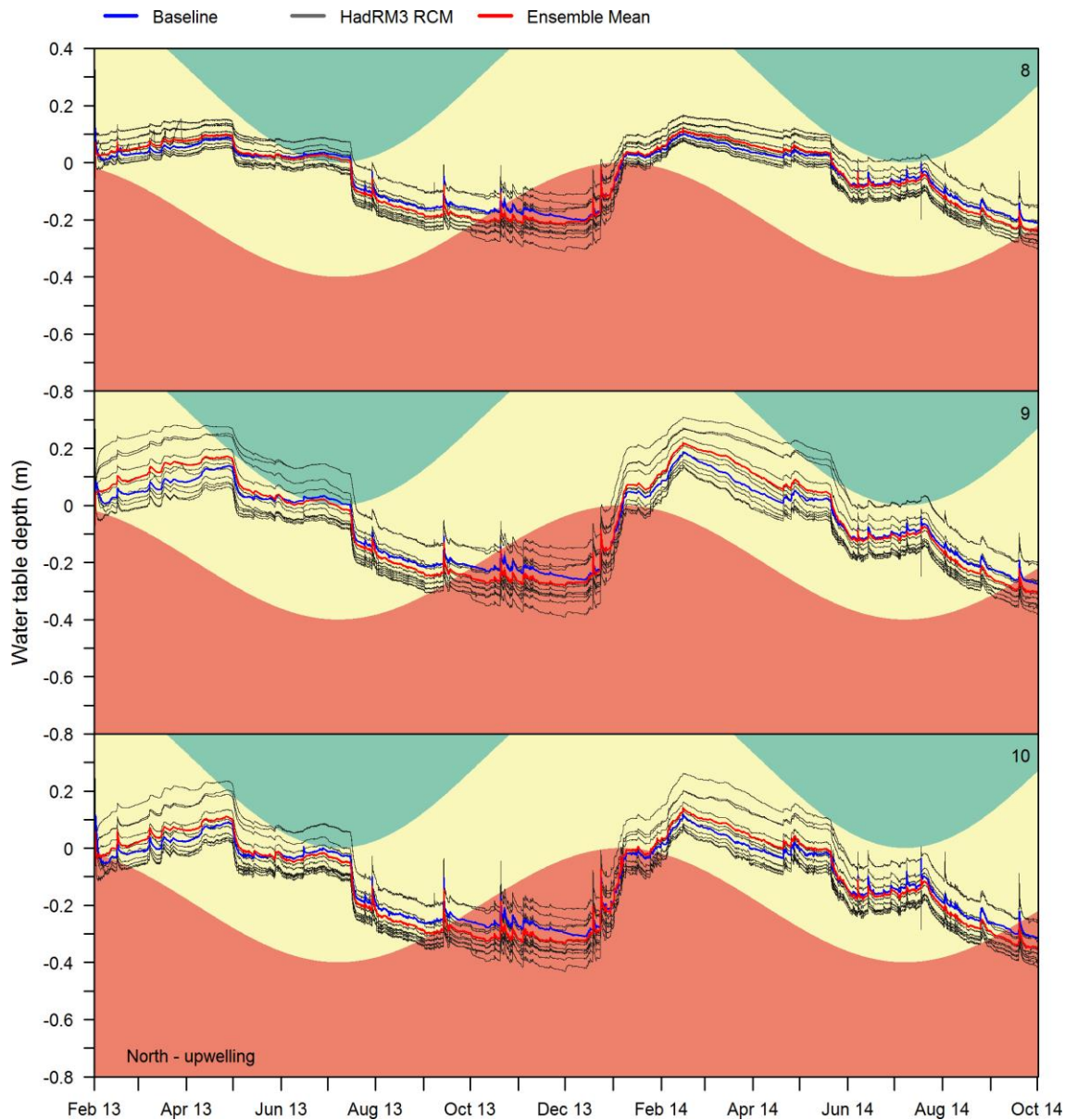


Figure 6-6. Simulated baseline, projected scenario member and mean wetland water table depths for North upwelling locations superimposed over the Desmoulin's whorl snail (*Vertigo moulinsiana*) water level requirements zone diagrams. Red - intolerable; Amber - tolerable for limited periods; Green – desirable

6.3 Assessment of fluvial ecological impacts of climate change

6.3.1 Physical habitat modelling

To assess physical habitat, depth and velocity characteristics of the River Lambourn were converted to habitat availability metrics using Habitat Suitability Indices (HSI)

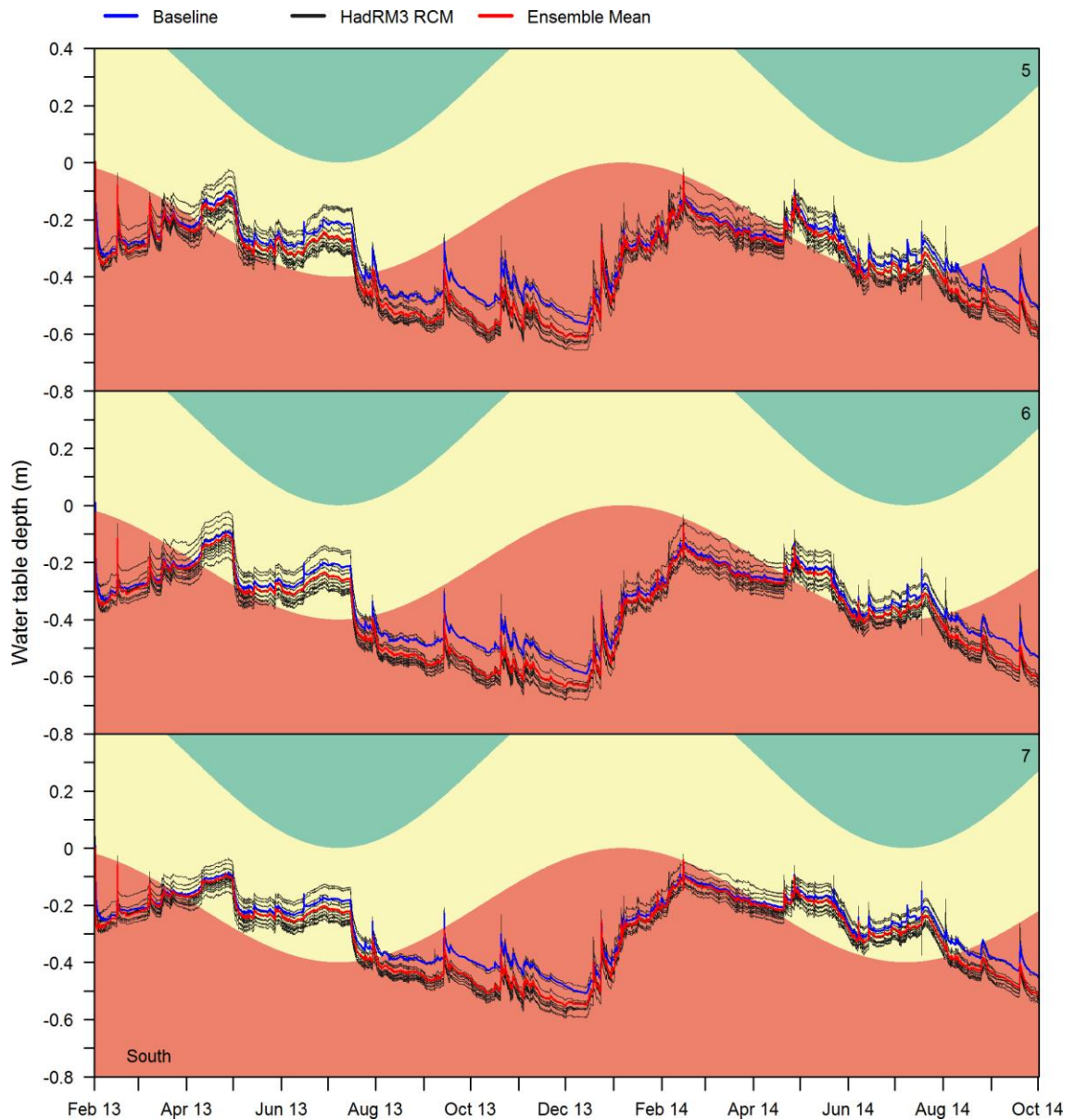


Figure 6-7. Simulated baseline, projected scenario member and mean wetland water table depths for South locations superimposed over the Desmoulin's whorl snail (*Vertigo moulinsiana*) water level requirements zone diagrams. Red - intolerable; Amber - tolerable for limited periods; Green – desirable

following the PHABSIM methodology (Bovee, 1982; Waddle, 2001). HSI for juvenile (0-7 cm) and adult (8-20 cm) brown trout (*Salmo trutta*) based on velocity and water depth were taken from Dunbar et al. (2001) (Figure 6-8). Bed substrate was not included as a parameter as the river has a uniform gravel bed.

Table 6-2. Percentage of full simulation period (01 Feb 2013 – 01 Oct 2014) simulated baseline, scenario and mean water levels are within each water depth zone (WDZ) for Desmoulin's whorl snail (*Vertigo moulinsiana*) at all piezometer locations. D, Desirable; T, Tolerable; I, Intolerable

Run ID	WDZ	North - no upwelling				North - upwelling			South		
		1	2	3	4	8	9	10	5	6	7
baseline	D	0.0	0.0	0.0	0.0	5.0	3.7	0.0	0.0	0.0	0.0
	T	58.1	61.8	39.3	59.7	82.1	78.8	75.0	35.0	32.7	44.3
	I	41.9	38.2	60.7	40.3	12.9	17.5	25.0	65.0	67.3	55.7
A	D	0.0	0.0	0.0	0.0	5.3	7.5	0.9	0.0	0.0	0.0
	T	64.8	68.6	51.2	64.7	81.8	75.3	77.6	33.1	31.8	42.0
	I	35.2	31.4	48.8	35.3	12.9	17.2	21.5	66.9	68.2	58.0
B	D	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	T	50.2	53.4	21.9	51.5	79.9	70.6	63.7	21.0	20.0	34.5
	I	49.8	46.6	78.1	48.5	20.1	29.4	36.3	79.0	80.0	65.5
C	D	0.0	4.9	0.0	4.0	12.4	22.6	8.8	0.0	0.0	0.0
	T	75.3	71.7	56.5	71.3	77.3	65.6	74.4	39.0	37.8	48.2
	I	24.7	23.4	43.5	24.6	10.3	11.8	16.8	61.0	62.2	51.8
D	D	0.0	6.4	0.0	4.9	12.5	24.3	9.9	0.0	0.0	0.0
	T	75.6	70.5	60.1	70.5	79.1	65.4	74.0	44.5	42.2	53.0
	I	24.4	23.2	39.9	24.6	8.4	10.4	16.1	55.5	57.8	47.0
E	D	0.0	0.0	0.0	0.0	7.0	17.0	3.5	0.0	0.0	0.0
	T	69.5	70.7	53.7	69.3	80.8	66.0	75.1	34.4	33.8	44.1
	I	30.5	29.3	46.3	30.7	12.1	16.9	21.3	65.6	66.2	55.9
F	D	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0
	T	56.4	59.2	36.4	56.7	81.9	78.1	70.2	23.1	22.7	35.9
	I	43.6	40.8	63.6	43.3	17.2	21.9	29.8	76.9	77.3	64.1
G	D	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	T	53.1	55.8	29.3	54.5	80.1	72.0	67.3	23.2	23.9	37.2
	I	46.9	44.2	70.7	45.5	19.9	28.0	32.7	76.8	76.1	62.8
H	D	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	T	53.4	54.7	26.7	53.7	81.9	76.3	67.4	22.4	22.1	35.2
	I	46.6	45.3	73.3	46.3	18.1	23.7	32.6	77.6	77.9	64.8
I	D	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	T	53.0	54.7	25.2	53.8	82.5	77.5	66.1	18.8	19.3	34.2
	I	47.0	45.3	74.8	46.2	17.5	22.5	33.9	81.2	80.7	65.8
J	D	0.0	0.0	0.0	0.0	2.6	0.2	0.0	0.0	0.0	0.0
	T	58.8	60.6	42.4	59.0	81.2	79.0	72.0	29.1	27.9	38.6
	I	41.2	39.4	57.6	41.0	16.2	20.7	28.0	70.9	72.1	61.4
K	D	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	T	51.0	53.8	25.3	52.3	79.2	71.7	64.7	25.5	24.3	36.2
	I	49.0	46.2	74.7	47.7	20.8	28.3	35.3	74.5	75.7	63.8
mean	D	0.1	0.1	0.0	0.1	5.5	3.6	1.1	0.0	0.0	0.0
	T	59.4	61.0	44.1	59.3	78.8	76.4	72.8	31.3	28.8	39.8
	I	40.5	38.9	55.9	40.6	15.7	20.0	26.1	68.7	71.2	60.2

The River Lambourn channel morphology was derived from a total of 41 cross-sections with an average spacing of 14.9 m, providing a total bed area 6735.1 m² and reach length 609 m (Figure 6-9). The distances between cross-sections provided the longitudinal lengths for a multi-dimensional matrix of cells with different bed areas and volumes, and, thus, hydraulic parameters (Figure 6-10).

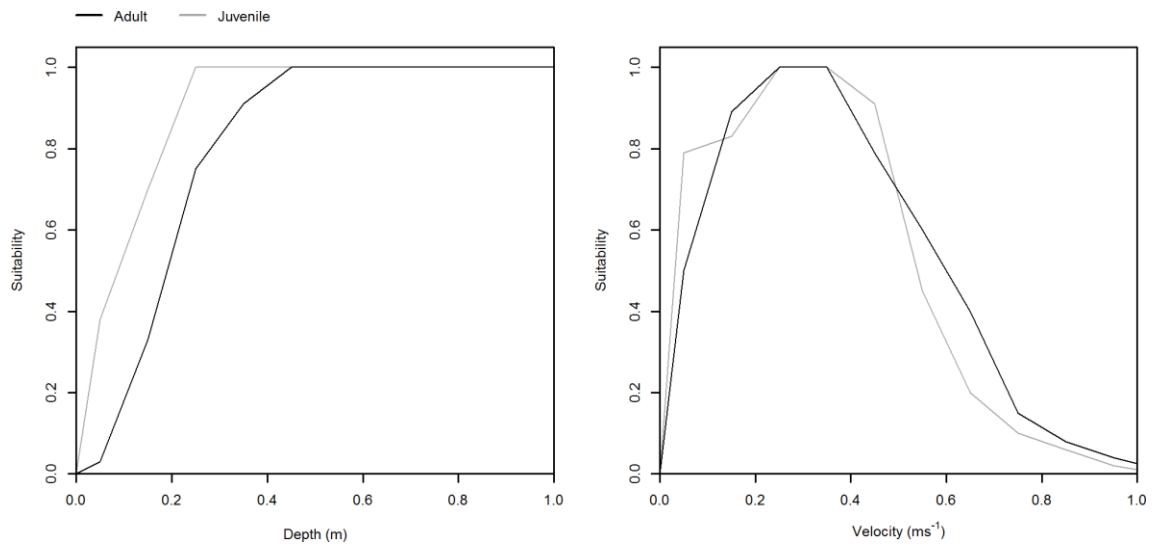


Figure 6-8. Habitat Suitability Indices (HSIs) for brown trout (*Salmo trutta*) (after Dunbar et al., 2001)

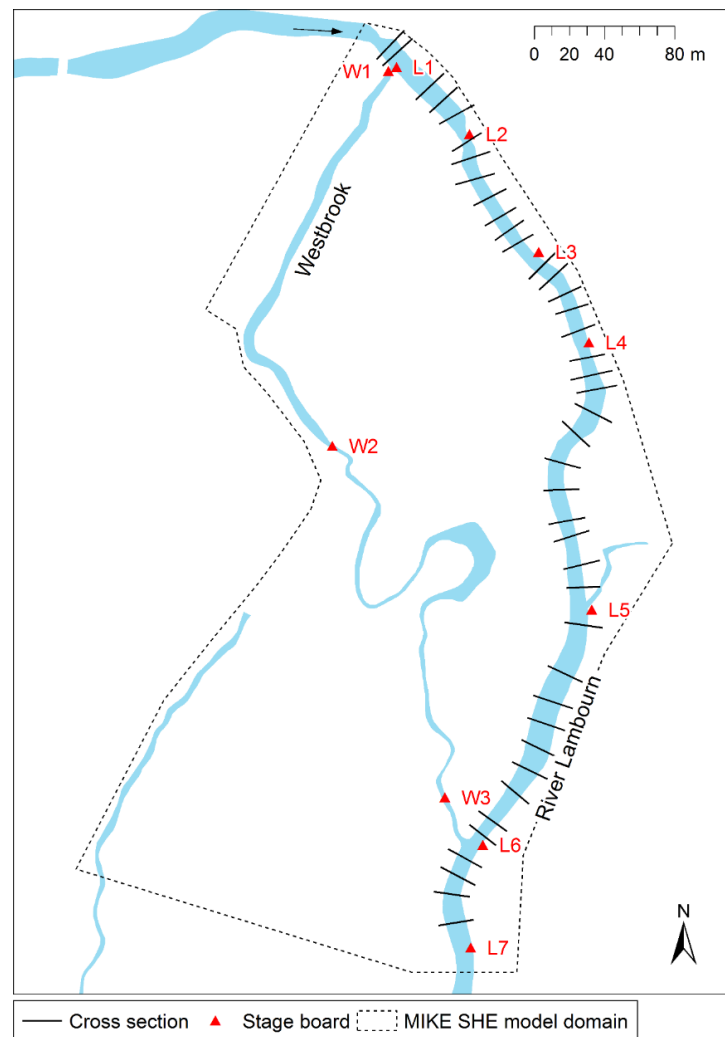


Figure 6-9. Map showing the locations of cross-sections for the physical habitat assessment, stage boards and MIKE SHE model domain

Transverse lengths were derived from changes in bed elevation from cross-section points, resulting in 641 computational cells. Since bed elevations are known from the cross-section survey, water depths for each cell were calculated from hydraulic model outputs of channel stage. The 1D velocity outputs were disaggregated to each cell by the ratio of cell flow area to total flow area for the cross-section. Depth and velocity for each cell were evaluated against the HSI and combined over the full range of discharges for the baseline, scenarios and scenario mean. These were totalled for the reach to produce available physical habitat, expressed as weighted usable area (WUA) in $\text{m}^2 \text{ 1000 m}^{-1}$ of river (Equation 6-1), as a function of discharge for the baseline:

$$\text{WUA} = \frac{\sum_{i=1}^n a_i v_i d_i}{l} \quad (\text{Equation 6-1})$$

where a_i is the surface area of cell i , v_i is the suitability associated with velocity for cell i , d_i is the suitability associated with depth in cell i , and l is the reach length in 1000 m (0.609). The slope of the output curve describes the physical habitat sensitivity to flow. WUA was also expressed as a time series for the baseline, individual scenarios and the scenario mean.

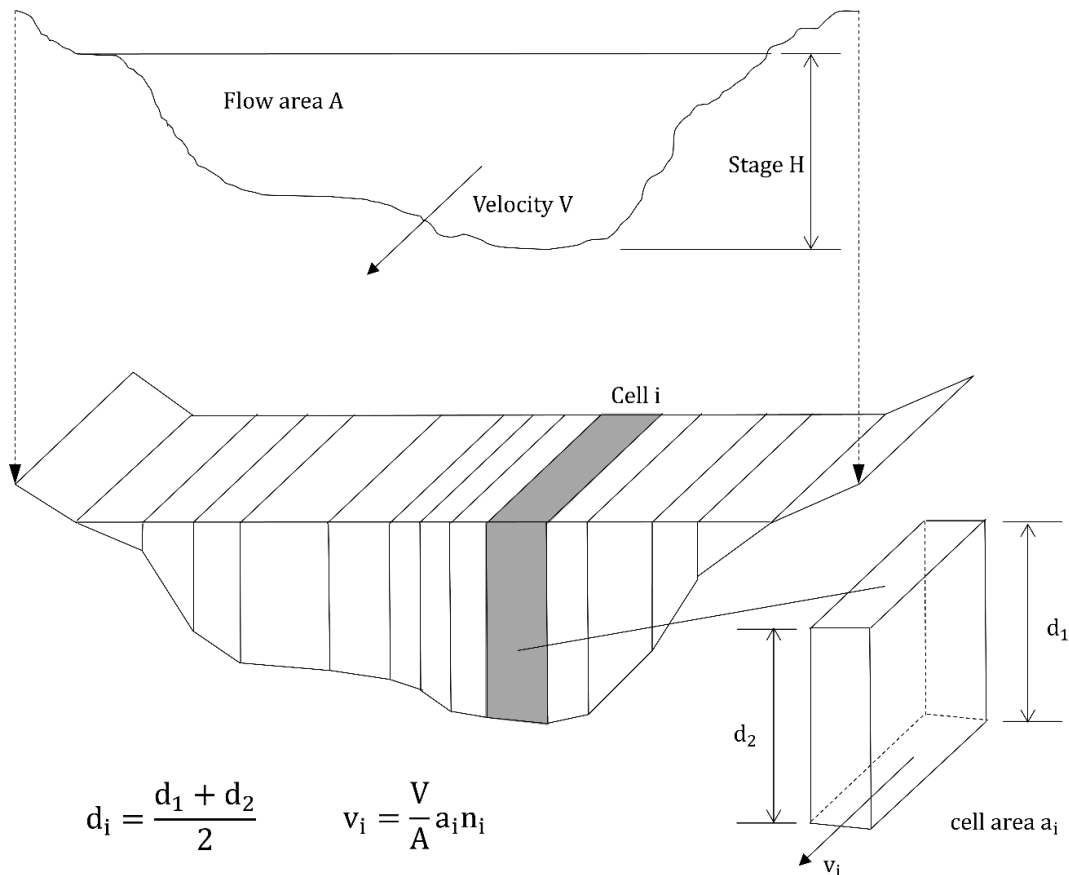


Figure 6-10. Schematic of habitat cell attribute matrix in PHABSIM (after Waddle, 2001)

To validate the model against observations, hydraulic measurements were taken from a field survey conducted between March and June 2015. Three distinct periods were identified to coincide with particular flow and vegetation conditions: (1) Winter high flows outside of the growing season, with minimal vegetation (10/3/2015 – 31/3/2015), (2) Summer low flows before a weed cut with abundant vegetation (7/5/2015 – 12/5/2015), and (3) Summer low flows after a weed cut with reduced vegetation (29/5/2015 – 8/6/2015). The weed cut took place on 13/5/2015. Velocity profiles and stage measurements were taken at each period for a total of 14 cross-sections (Figure 6-11). Cross-sections were split into vertical panels for measurement, the extents of which also provided the boundaries for the bed area cells used to calculate WUA. The study reach comprised a total bed area 4956.4 m² and length 488.3 m, reducing to 4514.1 m² and 442.6 m in period 2 due to high waters restricting accessibility. Discharge was measured using an electromagnetic flow meter. The same HSI for juvenile trout as that used in the

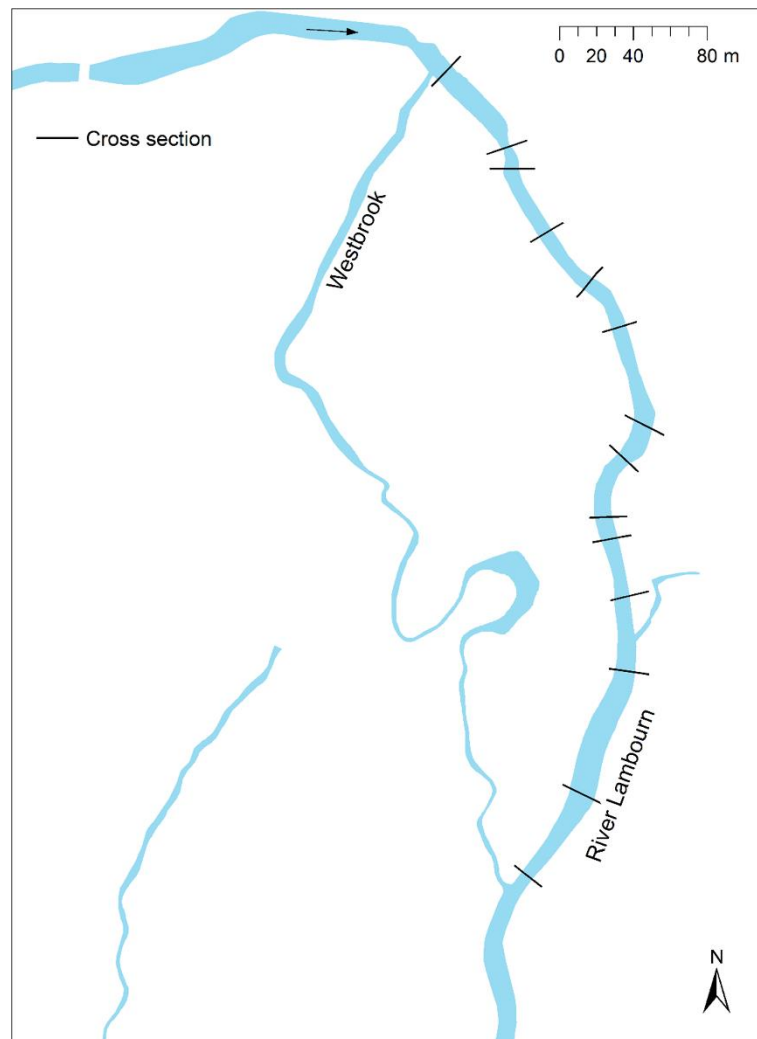


Figure 6-11. Locations of cross-sections for the field survey assessment of physical habitat availability (March – June 2015)

modelling study was applied to each cell to derive the proportion of available physical habitat. Values were summed according to Equation 6-1 to provide total WUA for the measured discharge. Results were standardised to percentage area cover for comparison with results from the 1D model based upon similar discharge and conditions.

6.3.2 Baseline flow and physical habitat characteristics

The influence of the regime on stage, velocity and, thus, available physical habitat is unclear, as there are no well-defined curvilinear relationships (Figures 6-12, 6-12 and 6-14), although the relationship between velocity and discharge does show a clear positive trend. The distribution of available physical habitat against discharge bears greater similarity to stage than velocity, although it is not a precise match. Overall, several values of stage, velocity and physical habitat exist for distinct discharges. This is most apparent below a flow of $4.15 \text{ m}^3\text{s}^{-1}$, corresponding to the 10% exceedance flow (Section 3.5.2), above which the flow duration curve steepens noticeably (Figure 3-15) and the point distributions for physical habitat, stage and velocity against discharge converge to closer relationships.

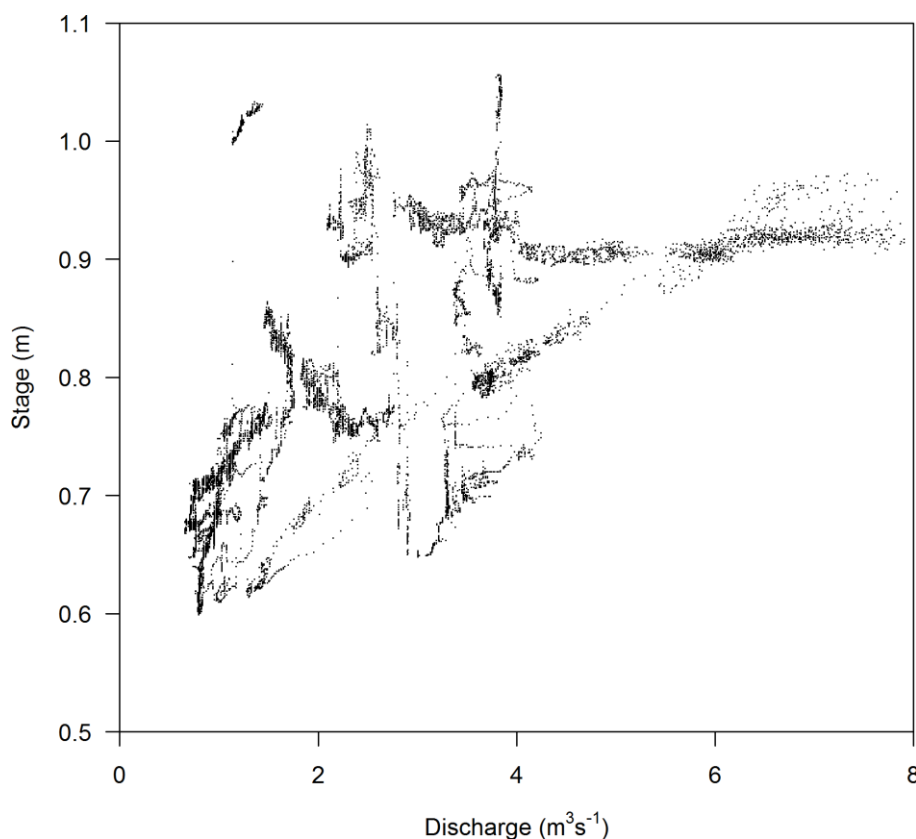


Figure 6-12. Relationship between flow and stage for the River Lambourn

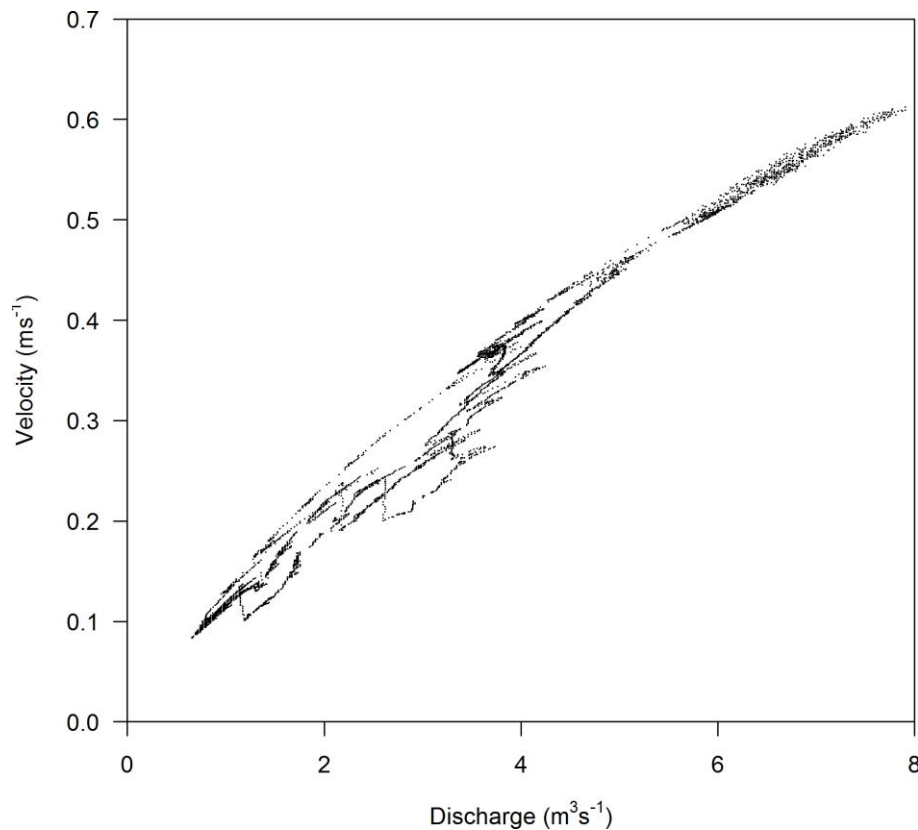


Figure 6-13. Simulated relationship between flow and velocity for the River Lambourn

The amount of available habitat appears greater for juvenile than adult brown trout at flows below $3.5 \text{ m}^3\text{s}^{-1}$. More habitat is available for adult trout when flows are between 3.5 and $5.0 \text{ m}^3\text{s}^{-1}$ whilst at discharges above $5.0 \text{ m}^3\text{s}^{-1}$ habitat availability for juveniles and adults is similar (Figure 6-13). Adult trout exhibit a greater range of habitat availability (6750.6 to $10685.6 \text{ m}^2 \text{ 1000m}^{-1}$) than juvenile trout (7513.8 to $10869.9 \text{ m}^2 \text{ 1000m}^{-1}$) from the WUA for the flows simulated. There is a greater difference between the minima ($763.2 \text{ m}^2 \text{ 1000m}^{-1}$) than maxima ($184.3 \text{ m}^2 \text{ 1000m}^{-1}$) for each life stage. Availability of baseline physical habitat varies considerably over the simulation period (Figure 6-14). Conspicuous peaks on and around 28/4/2013, 22/6/2013, 20/5/2014 and 20/7/2014 occur shortly before dramatic reductions caused by the weed cuts. The largest of these sudden drops on 1/5/2013 represents a decrease in available habitat of $1690 \text{ m}^2 \text{ 1000m}^{-1}$ (15.3%) for adult and $1790 \text{ m}^2 \text{ 1000m}^{-1}$ (16.2%) for juvenile trout. The period prior to this point was associated with both relatively high flow and high Manning's n (Figure 6-16). Another noticeable peak on 15/2/2014 corresponds to a period of high flow and is associated with the largest sustained increase in habitat that begins on 23/12/2013. The more gradual decline in physical habitat following the peak is due to declining discharge in the absence of a weed cut. Values of habitat availability and discharge during this period display much higher daily variability compared to any other

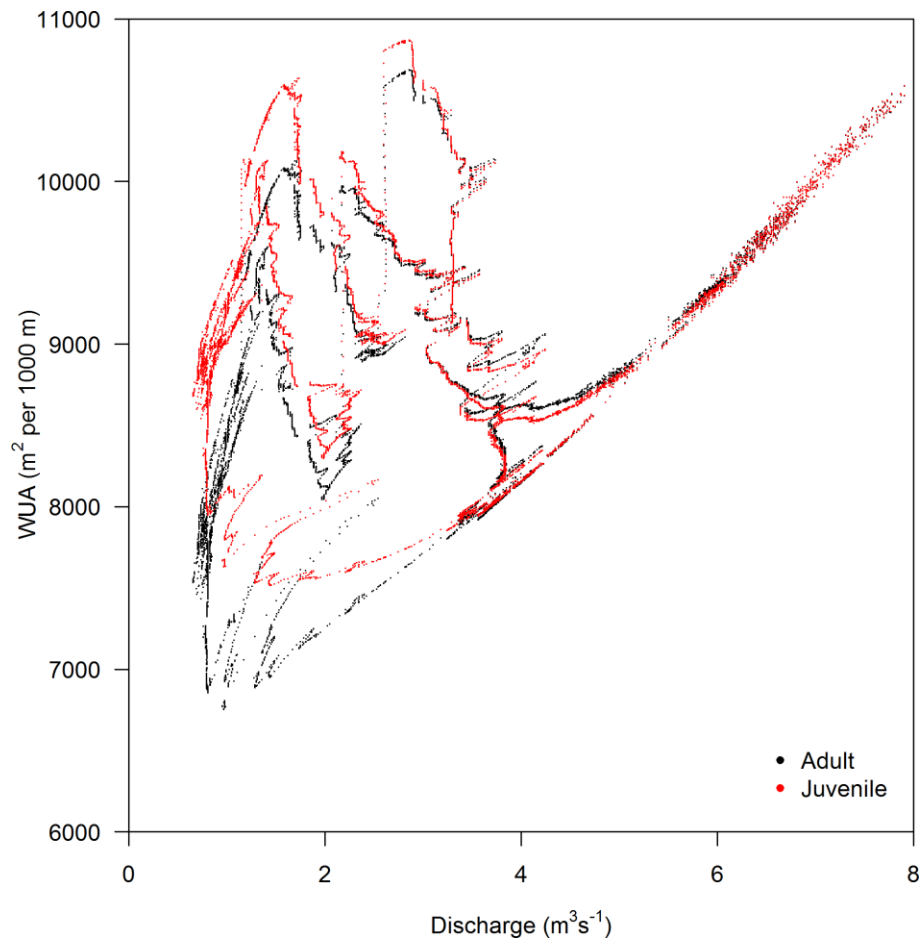


Figure 6-14. Simulated relationship between flow and physical habitat availability for brown trout (*Salmo trutta*) in the River Lambourn

time. Elevated available habitat match periods of high stage (see Sections 3.5.3 and 4.4.1).

Habitat availability for adult trout generally falls below that for juveniles. This difference is as much as 1148.5 m² 1000m⁻¹ (10.4%, 11/10/2013) during periods of low flow (e.g. July 2013 to February 2014 and August 2014 to October 2014). Contrasting periods of high flow display greater similarity in habitat availability of juvenile and adult trout and they are equivalent through April 2013 and January to March 2014. Available physical habitat is greater for adult trout in two periods, February 2013 to April 2013 and March 2014, although the differences (up to 95.3 m² 1000m⁻¹, 0.01%, 18/3/2013) are relatively small.

Values of WUA for juvenile trout from the field survey (Table 6-3) are plotted on Figure 6-14 at times with the same discharge conditions and stage in vegetation growth. To enable comparison, the percentage area cover of WUA is given on the secondary axis. For period 1, outside of the growing season and with a discharge of 1.77 m³s⁻¹, the

surveyed WUA was 69.2% of the area while the corresponding modelled WUA was 68.0% (1/1/2014). Period 2 (discharge of $1.26 \text{ m}^3\text{s}^{-1}$ prior to a weed cut in the growing season), resulted in 81.1 % of the area available as habitat for juvenile trout compared to 86.9% modelled (8/7/2014). Period 3 (discharge of $1.34 \text{ m}^3\text{s}^{-1}$ after a weed cut), resulted in 78.9% available habitat from the field survey and 82.4% from the model (2/8/2013). Although indicative only, the modelled and observed values are in good agreement. The model does over predict available habitat for similar discharges within the growing season, and under predicts the value outside of the growing season. However, differences, between 1.2% and 5.8%, are very small.

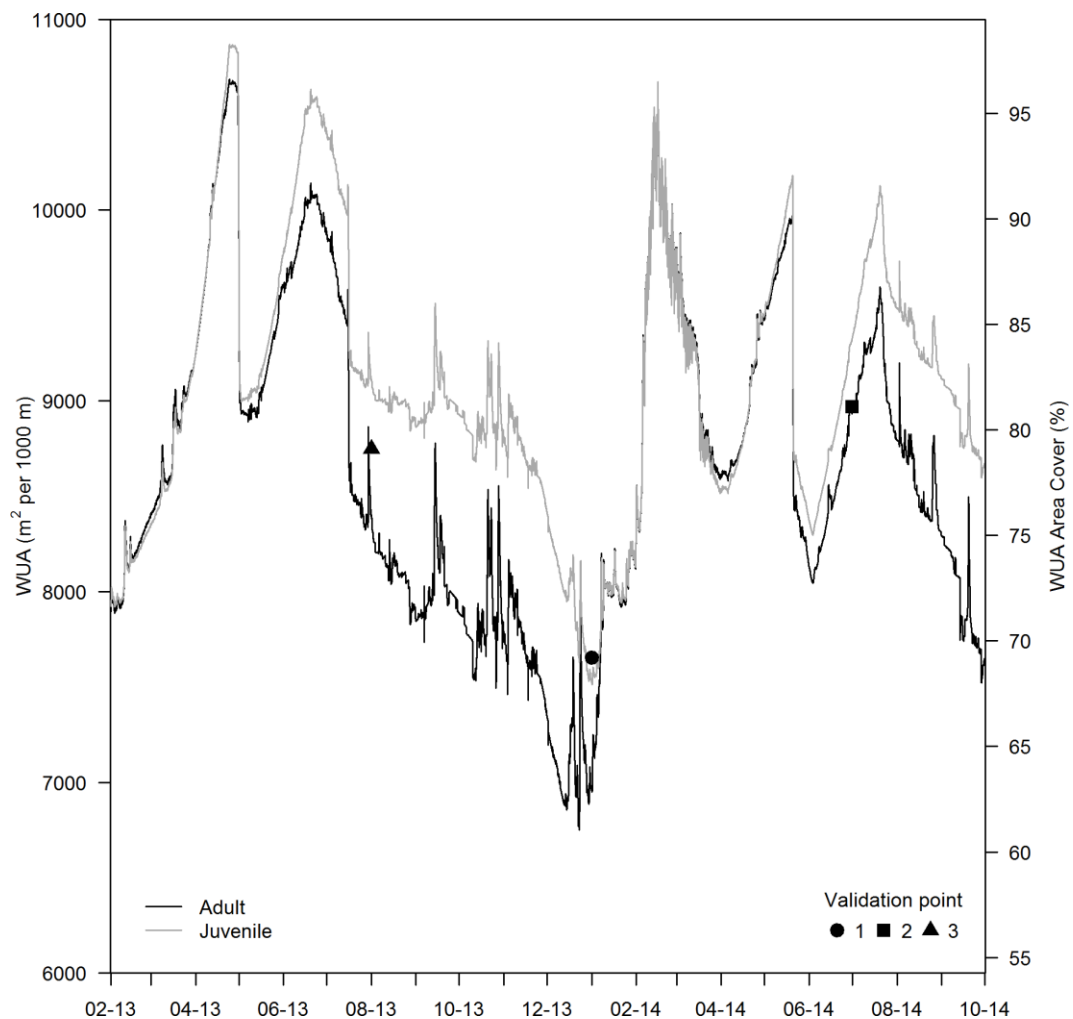


Figure 6-15. Simulated baseline physical habitat availability for adult (8-20 cm) and juvenile (0-7 cm) brown trout (*Salmo trutta*) and validation values from field survey periods for the River Lambourn: (1) Winter high flows outside of the growing season, with minimal vegetation (10/3/2015 – 31/3/2015), (2) Summer low flows pre-weed cut with abundant vegetation (7/5/2015 – 12/5/2015), and (3) Summer low flows post-weed cut with reduced vegetation (29/5/2015 – 8/6/2015)

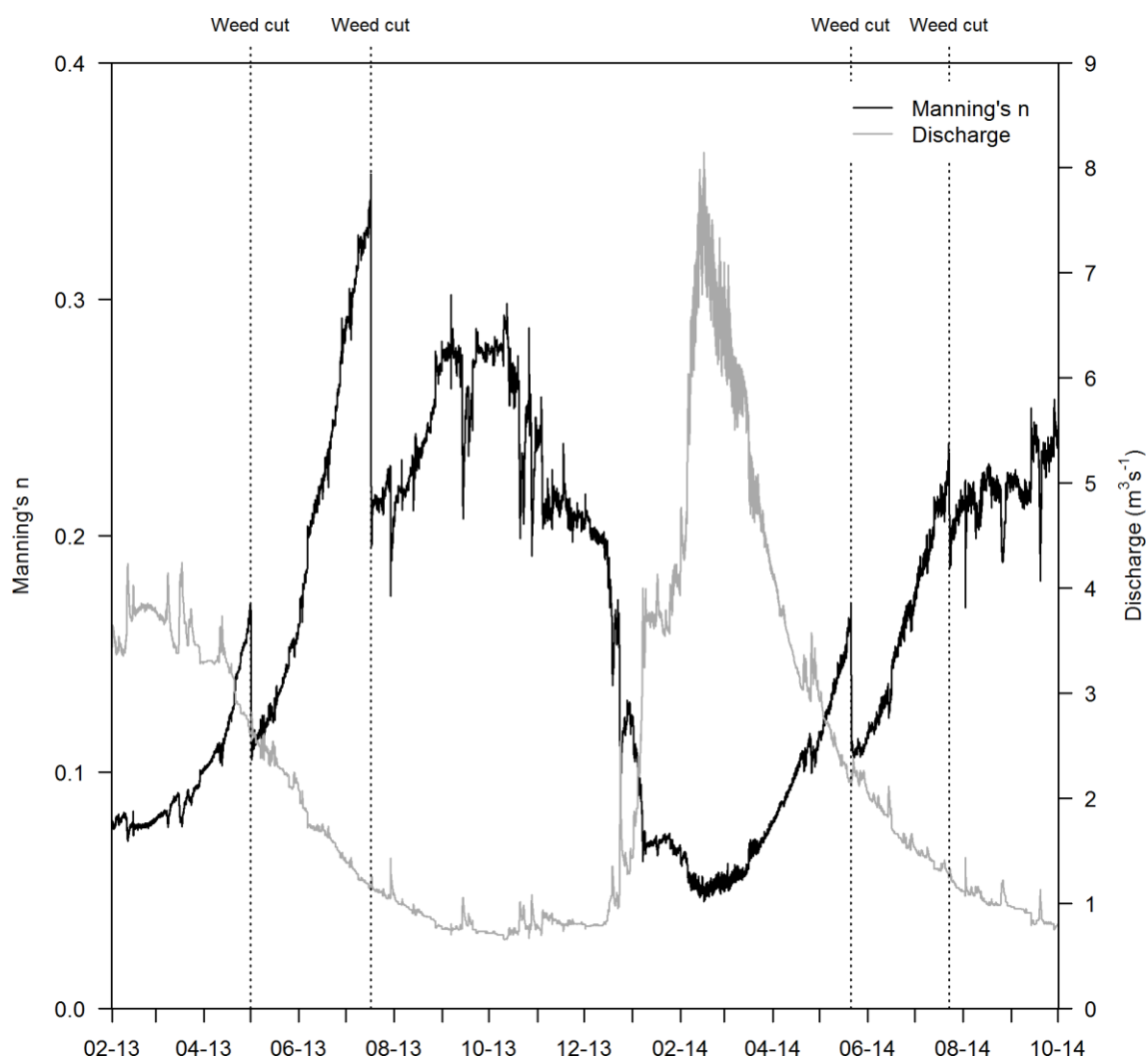


Figure 6-16. Manning's n roughness coefficient and discharge for the River Lambourn

Table 6-3. Validation conditions and physical habitat availability from the River Lambourn field survey (March-June 2015)

Period	Dates	Growing season	Pre/Post weed cut	Discharge (m^3s^{-1})	WUA (m^2 per 1000 m)	Bed Area (m^2)	Percentage cover (%)
1	10/3/2015-31/3/2015	N	N/A	1.77	7020.1	4956.4	69.2
2	7/5/2015-12/5/2015	Y	Pre	1.26	8273.7	4514.1	81.1
3	29/5/2015-8/6/2015	Y	Post	1.34	8005.8	4956.4	79.1

6.3.3 Climate change impacts on physical habitat availability

The impacts of the climate change scenarios on physical habitat availability is greater for adult rather than juvenile trout, especially during periods of low flow between July 2013 – December 2013 and August 2014 – October 2014 (Figure 6-17). This is highlighted by the scenario mean decrease averaged over the simulation period at $-349.4 \text{ m}^2 \text{ 1000 m}^{-1}$ (-4.0%) for adult and $-232.9 \text{ m}^2 \text{ 1000 m}^{-1}$ (-2.6%) for juvenile (Table 6-4). At no point over the full simulation period does the scenario mean show an increase in habitat availability for either life stage (Figure 6-17). The largest mean decreases are $-1305.0 \text{ m}^2 \text{ 1000 m}^{-1}$ for adult and $-871.2 \text{ m}^2 \text{ 1000 m}^{-1}$ for juvenile in October 2013. Closest correspondence with baseline values is seen in March 2013 at $-29.7 \text{ m}^2 \text{ 1000 m}^{-1}$ for adult and $-22.1 \text{ m}^2 \text{ 1000 m}^{-1}$ for juvenile.

Of the individual scenarios four (A, C, D, E) project increases in available habitat from baseline values on at least one day. For scenario A, this is only in April 2013 and April 2014, peaking at $+34.7 \text{ m}^2 \text{ 1000 m}^{-1}$ for adult and $+32.6 \text{ m}^2 \text{ 1000 m}^{-1}$ for juvenile. In contrast results for scenario D only drop below baseline values in October 2013 and then by at most $-17.3 \text{ m}^2 \text{ 1000 m}^{-1}$ for adult and $-13.3 \text{ m}^2 \text{ 1000 m}^{-1}$ for juvenile. The largest increases are projected by scenario D on 15/2/2014 at $+1303.2 \text{ m}^2 \text{ 1000 m}^{-1}$ for adult and $+1306.0 \text{ m}^2 \text{ 1000 m}^{-1}$ for juvenile. The largest inter-scenario range also occurs at this time, with scenario I projecting $-1007.5 \text{ m}^2 \text{ 1000 m}^{-1}$ for adult and $-1013.8 \text{ m}^2 \text{ 1000 m}^{-1}$ for juvenile. Of the other scenarios which display reductions in physical habitat, the greatest single decrease is for K on 3/11/2013 at $-2243.8 \text{ m}^2 \text{ 1000 m}^{-1}$ for adult and $-1553.2 \text{ m}^2 \text{ 1000 m}^{-1}$ for juvenile. The smallest inter- scenario range occurs in mid May 2013 for adult trout, ranging from -267.4 (I) to $+199.3 \text{ m}^2 \text{ 1000 m}^{-1}$ (D), and the beginning of June 2014 for juvenile trout, from -144.4 (I) to $+111.6 \text{ m}^2 \text{ 1000 m}^{-1}$ (D).

6.3.4 The influence of macrophyte growth on the habitat-discharge relationship

Physical habitat modelling approaches, such as PHABSIM, are widely used for research and regulatory purposes (Acreman et al., 2005; Dunbar et al., 2002; Reiser et al., 1989). Yet these assume the relationships between discharge and hydraulic characteristics are time invariant. The slope of the resulting physical habitat-discharge curve is used to indicate sensitivity to change in flow (Acreman et al., 2008). However, in watercourses where these relationships are disrupted, such as with groundwater interaction, morphological change, management practices or, here, weed growth, the efficacy of the method is diminished. Such factors are not, for example, accounted for by the inbuilt 1D hydraulic modelling protocols implemented within PHABSIM software packages (Waddle, 2001). The approach assumes a steady-state flow regime that is

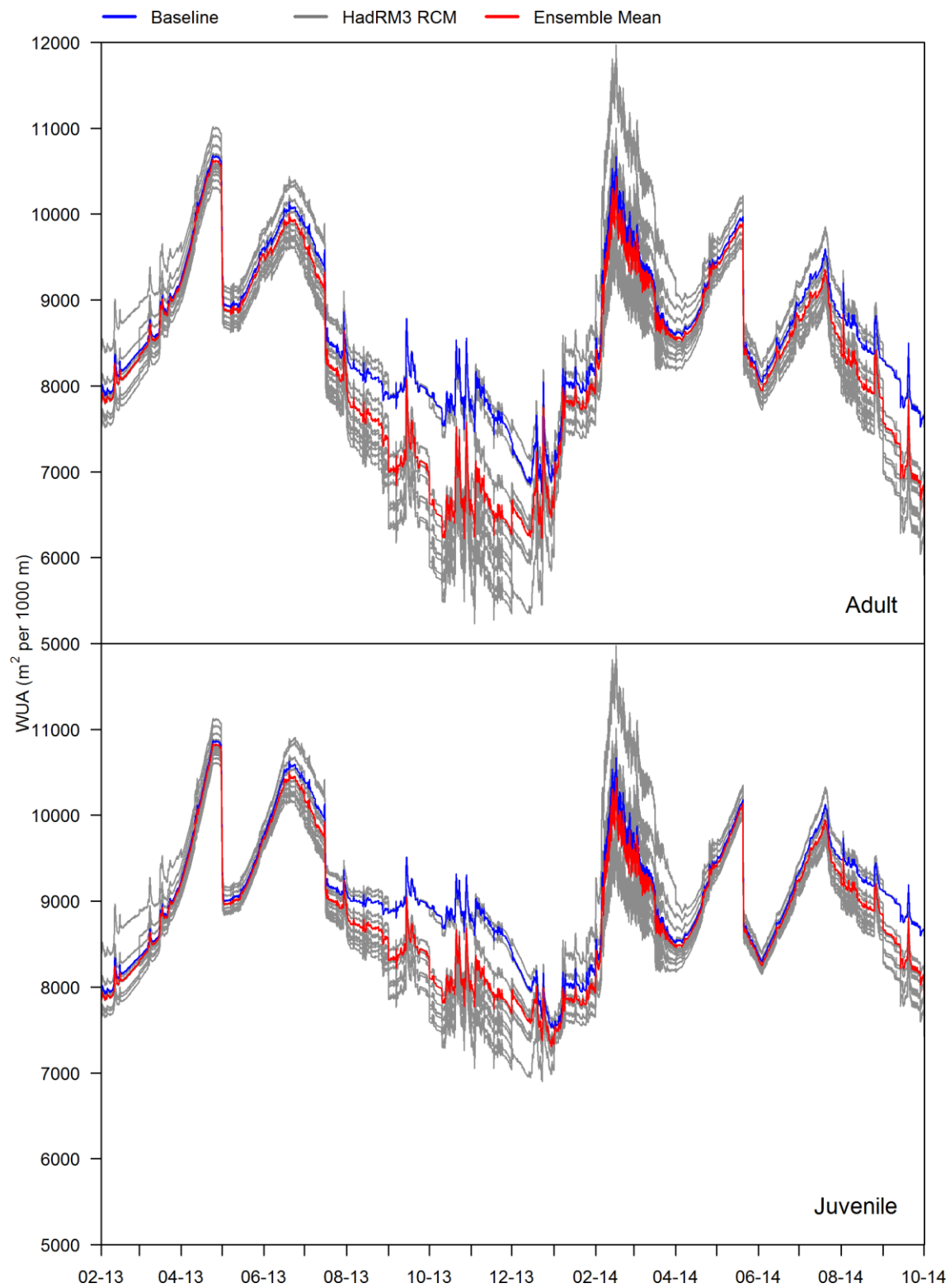


Figure 6-17. Simulated baseline, projected scenario and mean physical habitat availability for adult (8-20 cm) and juvenile (0-7 cm) brown trout (*Salmo trutta*) in the River Lambourn

*Table 6-4. Simulated baseline mean River Lambourn physical habitat availability (m^2 1000m⁻¹) for adult (8 – 20 cm) and juvenile (0-7 cm) brown trout (*Salmo trutta*) and changes for scenarios and mean for the full simulation (1/2/2013 – 1/10/2014) period. Italicised values indicate negative changes*

Run ID	Adult	Juvenile
baseline	8639.8	9071.8
A	<i>-308.0</i>	<i>-204.7</i>
B	<i>-681.3</i>	<i>-468.8</i>
C	141.8	116.4
D	308.8	260.0
E	<i>-186.6</i>	<i>-116.2</i>
F	<i>-583.1</i>	<i>-392.4</i>
G	<i>-558.5</i>	<i>-372.0</i>
H	<i>-634.6</i>	<i>-436.8</i>
I	<i>-640.2</i>	<i>-445.5</i>
J	<i>-386.2</i>	<i>-258.3</i>
K	<i>-596.2</i>	<i>-410.7</i>
mean	<i>-349.4</i>	<i>-232.9</i>

not applicable to the dynamic nature of many watercourses. The influence of macrophyte growth on stage-discharge (Figure 6-12), velocity-discharge (Figure 6-13) and physical habitat-discharge (Figure 6-14) relationships precludes their use to assess impacts of environmental change on habitat availability.

Output time series from the MIKE 11 1D hydraulic model of the River Lambourn, coupled to the MIKE SHE model of the Observatory, have allowed the effects of macrophyte growth and its management to be incorporated into an assessment of climate change impacts on physical habitat availability. Although this has been demonstrated for only a single species, brown trout, the method may be applied to other species for which HSIs are available, including other salmonids (Dunbar et al., 2001) and macroinvertebrates (Gore et al., 1998). Rather than using a single rating curve, the area of available physical habitat has been calculated at each time step over the full simulation period for baseline conditions along with each climate change scenario and the scenario mean. This allows a more flexible approach that is more in line with common hydrological/hydraulic assessments of climate change (e.g. Chun et al., 2009; Thompson, 2012; Thompson et al., 2009), and opens the way for similar use of models that are able to represent dynamic systems.

6.4 Spatial and temporal variations in ecological response to climate change

6.4.1 Wetland ecological response to climate change

A comparison of the water level requirements for the MG8 vegetation community and baseline water levels simulated by the MIKE SHE model explains why only remnants of this community are currently found in the south meadow of the Lambourn Observatory. Current conditions are infrequently desirable and often stray into the dry intolerable range. The seasonal pattern of water level lows and highs, characteristic of the area, does not match with the requirements for each species and would contribute to their current decline. It is realised that the water level requirements provided by the literature (Wheeler et al., 2004) indicate the broad range of hydrological regime that gives rise to specific vegetation communities. Nevertheless, for the MG8 community there is detailed data to identify the magnitude of hydrological impact that would have an effect. Hence, the requirements are considered robust. Model results suggest that climate change is likely to push wetland water levels further into intolerable conditions that would further facilitate succession by other communities. An extended duration of waterlogging, as seen for scenarios A, C, D and E, would cause the community composition to change from grassland to mire or swamp (Gowing et al., 2002; Wheeler et al., 2004). Conversely, deeper water tables, as simulated for the other scenarios, would cause a gradual loss of characteristic, moisture demanding species such as marsh marigold (*Caltha palustris*), ragged robin (*Lychnis flos-cuculi*) and common spikerush (*Eleocharis palustris*). Waterlogging already occurs in the north meadow, especially around groundwater upwelling areas, where the communities S5 *Glyceria maxima* swamp and S6 *Carex riparia* swamp are prevalent (Section 3.9.2). Any increase in water levels is likely to cause an expansion of these swamp areas. Declines of the magnitude simulated by some of the climate change scenarios (I, K) would likely have little restorative effect in upwelling areas, and in the rest of the north meadow may cause drying out during late summer and the consequent loss of wetland species in favour of tall-herb communities such as S28 *Phalaris arundinacea* tall-herb fen, OV24 *Urtica dioica*-*Galium aparine* and OV26 *Epilobium hirsutum*. In the south meadow, where remnants of MG8 still exist, the community has a better chance of recovery and expansion. Predicted levels for the climate change scenarios are on the whole within desirable or tolerable ranges, only falling outside of these when projections result in decreases through the late summer and early winter months. Management efforts for the MG8 community would therefore likely to be more productive when directed towards the south meadow.

Hydrological requirements for Desmoulin's whorl snail obtained from the literature (Tattersfield and McInnes, 2003) are uncertain, but do provide some indicative water levels which can be used to assess the potential impacts of climate change on this individual species. Survival of the snail is dependent on the maintenance of high water levels and standing water. It is clear that the areas where the snail is most likely to survive at the site are around the zones of upwelling in the north meadow. However, even in these areas and under the extreme climate change scenario, water levels rarely reach elevations that are considered desirable. Those climate change projections associated with lower water levels result in the creation of periods of intolerable hydrological conditions, even in these relatively wet areas. In the south meadow, where the simulated water levels do not exceed ground level under the baseline and any scenario, conditions are unlikely to support any Desmoulin's whorl snail.

6.4.2 Fluvial ecological response to climate change

Adult brown trout are more susceptible to low flows than juvenile trout. At high flows the simulated differences in habitat availability between the two life stages become less apparent. Periods of low flow are also associated with an amplified response of available physical habitat to climate change in most scenarios. The scenario mean shows an overall reduction in habitat availability; however, the responses reflect some uncertainty in the projected hydrometeorological drivers for discharge inputs to the MIKE 11 model. Inter-scenario member variations differ over time and exhibit seasonality in the nature of these changes. An existing vulnerability of brown trout during low summer flow periods is exaggerated in projections from nine of the 11 scenarios, especially for the adult life stage. These nine members project general reductions in habitat availability through the full simulation period. In the other two scenarios (C and D), there is an overall increase in habitat availability throughout the simulation period although it is close to baseline values during periods of low flow. Nevertheless, without modifications to current management practices, the available physical habitat for brown trout would decline under the majority of climate change scenarios.

As indicated in Section 5.5, variations in stage are more pronounced than for velocity between different scenarios. Climate induced changes in habitat availability for brown trout will be depth limited by the hydraulic geometry. In addition, stage appears to be the dominant factor in controlling the baseline amount of available habitat. This is expressed in the similarity of the stage-discharge relationship to the physical habitat-discharge relationship (Figures 6-12 and 6-14). Through much of the simulated reach of the River Lambourn, velocities are generally within the optimal or near optimal HSI range for brown

trout, only falling outside during periods of extreme low or high flow. Stage values largely fluctuate through optimal and suboptimal HSI ranges. Stretches with relatively low stage and higher, more variable velocities (L1, L3-5), most likely representing fast runs, are likely to be more impacted by climate change. Deeper sections of the river in which velocity is more stable, such as pools, may provide refuge areas for brown trout during periods of stress induced by reduced flow. However, spatial differences in vegetation are not included in this study. Assumption of a single Manning's n value over the reach, plus disaggregation of velocity profiles across cross-sections, does not allow for local small-scale variations (microhabitat) (Sutcliffe, 2014) and renders such a spatial evaluation inappropriate. An assessment of this order would require a field survey before and after a defined change in conditions, such as a weed cut. On the reach scale, however, where these variations average out, and validation to stage is excellent, the method provide a useful means of assessing the impacts of climate change.

Validation values for available physical habitat, although close to those modelled, are indicative only. The field survey was undertaken the year after the simulation period, and encompasses a section of the modelled reach with a different area and fewer cross-sections. An extension of the model simulation period to incorporate the field survey period was not possible due to availability constraints on meteorological and groundwater level data. Nevertheless, both sets of results exhibit the rise and fall in habitat availability due to the macrophyte growth season and its management. When compared at times with similar conditions and identical flows, the values are reassuringly close.

6.5 Summary

To assess the ecological impacts of climate change in the wetland, simulated water levels were linked to requirements of the MG8 plant community and Desmoulin's whorl snail (*Vertigo moulinsiana*) for which the Observatory is designated. Impacts on each differed spatially and in line with hydrological impacts. For the MG8 vegetation community current conditions are infrequently desirable, explaining why only remnants of this community are currently found in the south meadow. Under climate change, and especially in the north meadow, wetland water levels are likely to extend further into intolerable conditions that would facilitate succession by other communities. In the south meadow, predicted levels under the climate change scenarios are generally within desirable or tolerable ranges. Here, where remnants of MG8 still persist, the community has a much better chance of recovery and expansion. Desmoulin's whorl snail, on the other hand, is most likely to survive at the site around the zones of upwelling in the north

meadow where the high water levels and standing water necessary for its survival are likely to continue. In the south meadow, where water levels are not projected to exceed ground level under any scenario, conditions are inauspicious for the snail.

In the River Lambourn, an assessment of physical habitat availability for brown trout revealed an overall reduction induced by climate change, more so for adult rather than juvenile brown trout. Impacts accentuated periods of low flows during summer months. Assessment of physical habitat availability for brown trout over the model simulation period revealed the projected impacts of an ensemble of climate change scenarios. Impacts were most pronounced during summer months accentuating periods of low flows and reduced habitat. An overall reduction in habitat availability was larger for adult rather than juvenile brown trout. Model results demonstrated the effects of weed cutting on flow depth, velocity and habitat availability. Indeed, the influence of macrophyte growth and its management on stage and velocity caused the physical habitat-discharge relationship itself to be unusable in evaluating the sensitivity of brown trout to changes in flow. The application of the MIKE 11 hydraulic component of the MIKE SHE model revealed the variability in flow conditions and, thus, habitat availability for the river.

The differences in the water level requirements of the MG8 community and Desmoulin's whorl snail suggests that there is some potential to manage the wetland for the promotion of each in different parts of the site. There is also the potential to rethink the management regime for macrophytes in order to mitigate the impacts of changes to the flow regime, such as through climate change. Managing for multiple objectives is an important consideration for areas, such as the Lambourn Observatory, where complicated feedbacks between hydrology and biotic communities exist. Such multiple objective management provides the focus for the following chapter.

Chapter 7

Discussion of management implications

7.1 Introduction

Wetland management practices revolve around manipulating water levels and flow to meet regulatory requirements and/or maintain processes and services such as the conservation of desired species or communities, flood mitigation, water quality and supply, and arable or pastoral productivity (Morris et al., 2008). Currently at the Lambourn Observatory, management is centred on cutting back instream macrophyte growth to increase flood conveyance, reduce riparian water levels and maintain fisheries (Old et al., 2014). However, wetland species for which the site is designated are in decline (Natural England, 2012), and without modifications to current management practices the research undertaken in this study suggests that ecological conditions would worsen under the majority of climate change scenarios (Chapter 6). Hydrological conditions and, in turn, the response to climate change differ noticeably over relatively short distances through the Observatory (Chapter 5). Differences in the hydrological requirements of designated and productive species at the site indicate the possibility of multiple objective management. Management efforts could focus on different species in distinct areas of the site.

In this chapter, the ecohydrological impacts of the current management regime and the implications of climate change induced alterations are examined for designated and productive species at the Lambourn Observatory. Recommendations are suggested for adaptive management actions, developed from modifications to the existing management practices, and spatially targeted to the requirements of each species.

7.2 The effects of the current management regime

Results of the field monitoring programme (Chapter 3), replicated and augmented by the MIKE SHE model results (Chapter 4), demonstrate that instream weed cutting has profound effects upon the Observatory's existing hydrological processes and ecology. Weed cuts are typically carried out to increase flood conveyance, reduce riparian water levels and maintain fisheries (Baatrup-Pedersen and Riis, 2004; Nikora et al., 2008; Old et al., 2014). At the Observatory, under the Flood and Water Management Act 2010, CEH must fulfil legal obligations as the riparian landowner to maintain the conveyance of river water through the site to manage flood risk. The weed cutting undertaken to achieve this is carried out under the advice of a downstream river keeper, whose main

priority is to maintain fish habitat. Dependent on seasonal flows, weed cuts are carried out twice or three times each year. Approximately 40-50% of the weed is removed over a day by manual cutting to leave a sinuous flow pattern (Figure 7-1).

Implications of weed cutting at the Observatory have been previously described to some extent by Old et al. (2014). Three weed cuts on 9/7/2008, 20/5/2009 and 5/5/2010, were found to result in a number of hydrological effects for both instream and riparian environments. Although relatively small (3-9%) changes in discharge of the River Lambourn were observed, removal of weed caused drops in stage of 22% in 2008, 28% in 2009 and 17% in 2010 (Table 7-1). Mean cross-sectional velocities increased by over 40%, with equivalent decreases in Manning's n , while increases in conveyance capacity ranged from 89% to 141%. In the wetland, observations were based on a transect of selected piezometers from the Atkins (2005) instrumentation network (Figure 7-2). The sharp decrease in channel stage at the boundaries of the transect signalled a rapid response in wetland water levels. Gravel levels dropped rapidly within the first 24 hours, with peat levels falling at a slightly slower rate. Wetland water levels stabilised at relatively constant values after 72 hours.

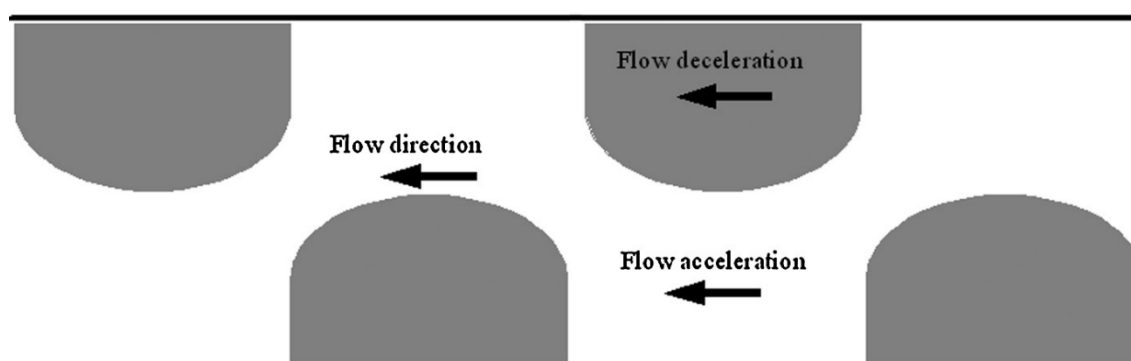


Figure 7-1. Weed cutting pattern to leave sinuous flow (after Old et al., 2014)

Table 7-1. Flow conditions before (BC) and after (AC) weed cuts on 9/7/2008, 20/5/2009 and 5/5/2010 (after Old et al., 2014). Percentage change in parentheses

Event		Discharge (m^3s^{-1})	Water level (m)	Cross-sectional mean velocity (ms^{-1})	Manning coefficient (n)	Conveyance capacity (m^3s^{-1})
2008	BC	2.08	1.07	0.30	0.13	2.08
	AC	2.03 (-3%)	0.84 (-22%)	0.43 (+43%)	0.08 (-43%)	3.82 (+89%)
2009	BC	1.57	1.06	0.22	0.18	1.68
	AC	1.46 (-7%)	0.77 (-28%)	0.35 (+55%)	0.08 (-54%)	3.52 (+141%)
2010	BC	2.04	0.92	0.36	0.10	3.03
	AC	2.22 (+9%)	0.77 (-17%)	0.52 (+45%)	0.05 (-45%)	5.21 (+135%)

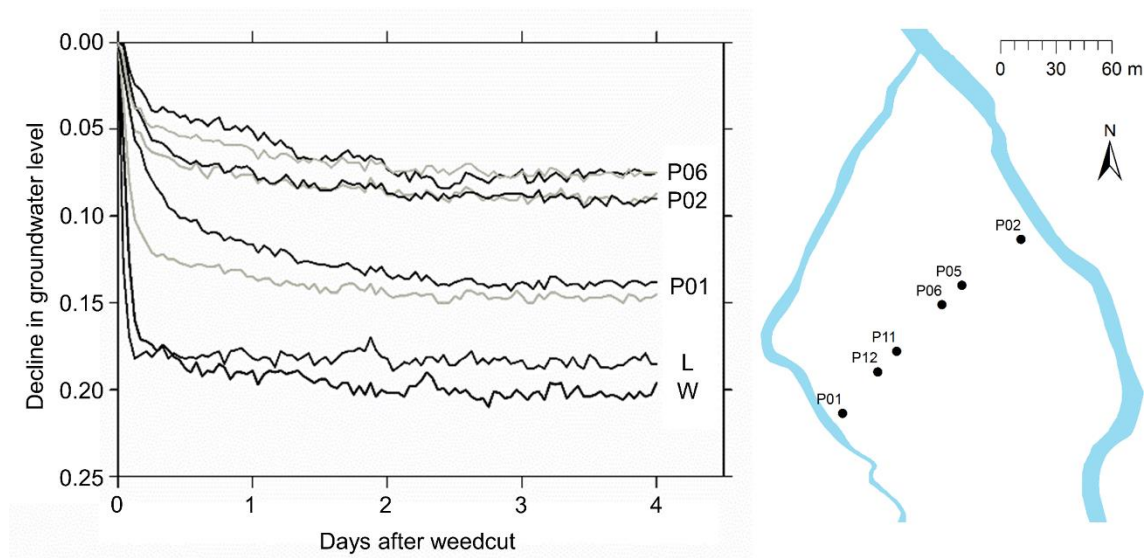


Figure 7-2. Water level responses following a weed cut on 5/5/2010 and locations of piezometers from the Atkins (2005) network (after Old et al., 2014). Peat (black) and gravel (grey) data from paired piezometers at P01, P02 and P06. Lambourn (L) and Westbrook (W) data from stage readings in line with north transect

Findings from this study support those of Old et al. (2014), with sharp observed and simulated decreases in stage and small increases in velocity, yet relatively little change in discharge, following weed cuts on 1/5/2013, 16/7/2013, 21/5/2014 and 23/7/2014 (Table 7-2). Changes for each variable are illustrated for the weed cut on 16/7/2013 (Figure 7-3). Drops in stage of -15%, -22%, -14%, -8% and increases in velocity of +25%, +40%, +26% and 8% were seen for cuts on 1/5/2013, 16/7/2013, 21/5/2014 and 23/7/2014 respectively. In the River Lambourn these hydraulic changes resulted in coincident rapid and relatively large reductions in habitat availability for brown trout. Periods of high flow and abundant macrophyte coverage were associated with large areas of available physical habitat, while at low flows following weed cuts habitat availability was at its lowest. Habitat availability reduced by -15%, -9%, -15% and -5% for adult brown trout and by -17%, -8%, -15% and -5% for juveniles on 1/5/2013, 16/7/2013, 21/5/2014 and 23/7/2014 respectively.

Aside from their hydraulic significance, submerged plants within water courses are generally seen as beneficial for fish due to the provision of habitat for invertebrate prey, shelter and protection, and oxygen production (Bursche, 1971). On the other hand, sediment may accumulate due to locally reduced velocities and deposits of detritus from dying plants, leading to a silting of the bed substrate. Higher velocities mobilise sediment and ensure that clean gravel spawning habitats are maintained (Soulsby et al., 2001), a principal motivation for weed removal at the Observatory. Nevertheless, juvenile

Lamprey, for which the site is designated, require soft sediment for habitat and food (Clemens et al., 2010).

The species-poor tall-herb fen and swamps prevalent in the wetland of the Lambourn Observatory reflect the duration and magnitude of drops in groundwater levels from weed cutting. Substantial drops in water level are seen, with the 16/7/2013 weed cut having the most impact, causing drops of -0.13 m in North – upwelling, -0.17 m in North –no upwelling and -0.18 m in South areas with variable effects (Table 7-2). Water levels fall into the lower intolerable zone for Desmoulin’s whorl snail, established in the north meadow; and, in the south meadow, where remnants of the MG8 vegetation community persist conditions approach intolerable (Figure 7-3v and 7-3vi). Later in the year and in the south meadow, the detrimental effects of low water levels on MG8 through summer and autumn are exaggerated due to the preceding spring and early summer weed cuts. Both the MG8 vegetation community, notable as a habitat for breeding snipe (*Gallinago gallinago*), and Desmoulin’s whorl snail, considered to be Near Threatened in Great Britain and on the IUCN red list of threatened species (Killeen et al., 2012), are currently in decline at the site (Natural England, 2012). Indeed, the meadows of the Observatory once supported breeding snipe (Everard, 2005), but these are no longer present and a general decline in the UK and Europe has been linked to losses in lowland wet grassland (Ausden et al., 2001).

Table 7-2. Discharge, stage and velocity at L2, physical habitat availability (WUA) for brown trout (Salmo trutta), and wetland water table depths at North – upwelling (piezometer 9), North – no upwelling (piezometer 2) and South (piezometer 5) locations before (BC) and after (AC) weed cuts on 1/5/2013, 16/7/2013, 21/5/2014 and 23/7/2014

Weed cut		Discharge (m ³ s ⁻¹)	Stage (m)	Velocity (ms ⁻¹)	WUA adult (m ² per 1000m)	WUA juvenile (m ² per 1000m)	North – upwelling (m)	North - no upwelling (m)	South (m)
01/05/2013	BC	2.62	1.04	0.20	10615.4	10807.4	0.14	0.04	-0.11
	AC	2.71	0.88	0.25	8996.1	8998.5	0.03	-0.09	-0.22
16/07/2013	BC	1.15	0.96	0.10	9396.3	9951.5	0.00	-0.10	-0.41
	AC	1.17	0.75	0.14	8581.6	9208.4	-0.13	-0.27	-0.26
21/05/2014	BC	2.19	0.92	0.19	9945.2	10153.0	0.02	-0.09	-0.23
	AC	2.19	0.79	0.24	8423.6	8684.3	-0.12	-0.25	-0.25
23/07/2014	BC	1.25	0.84	0.13	9498.0	10117.4	-0.07	-0.17	-0.28
	AC	1.29	0.77	0.14	8991.3	9633.6	-0.15	-0.28	-0.32

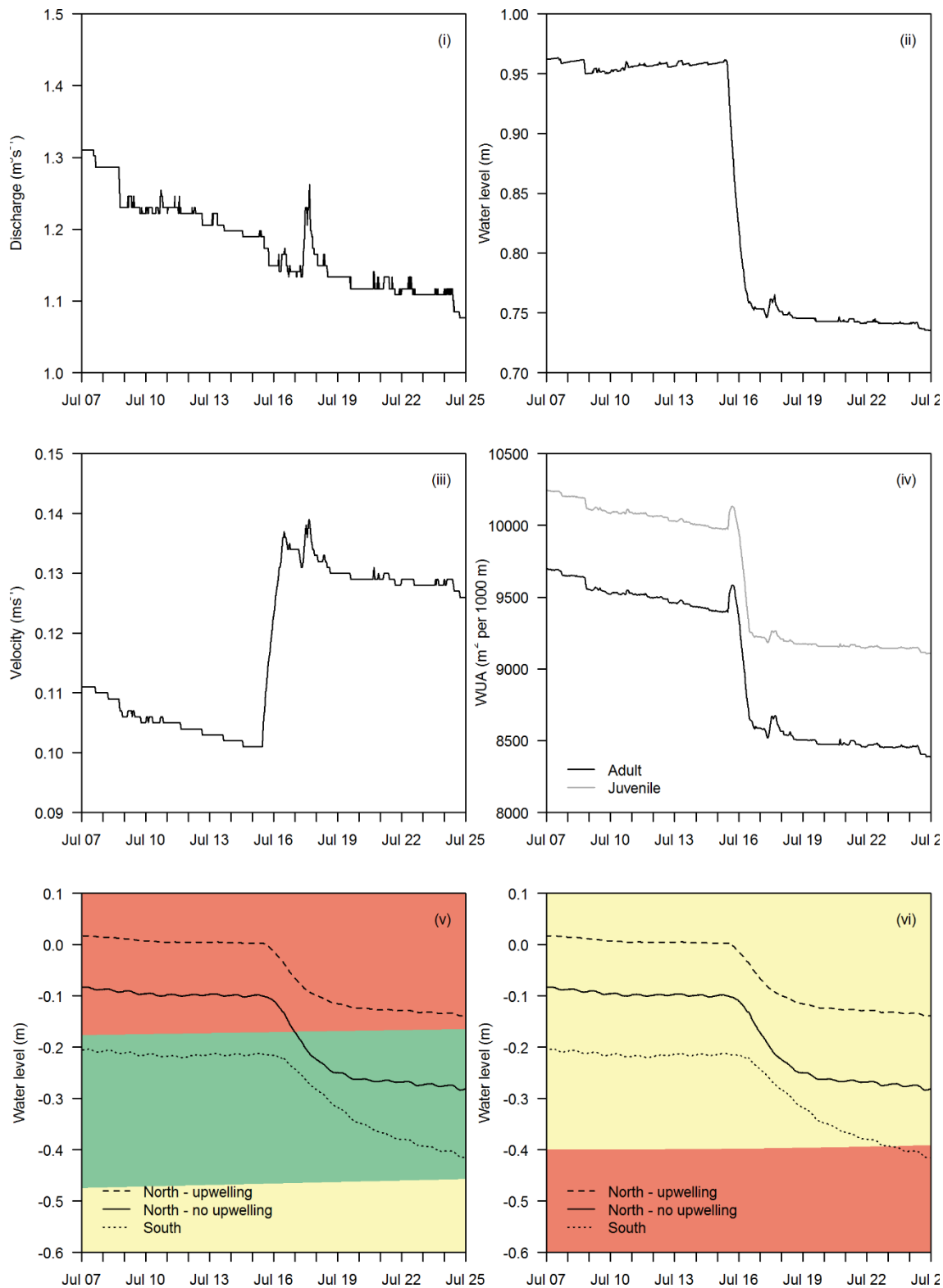


Figure 7-3. Impacts of the 16/7/2013 weed cut on: (i) discharge at L2, (ii) stage at L2, (iii) velocity at L2, (iv) physical habitat availability (WUA) for adult and juvenile brown trout (*Salmo trutta*), and wetland water table depths at North – upwelling (piezometer 9), North – no upwelling (piezometer 2) and South (piezometer 5) locations superimposed over water level requirements for (v) MG8 vegetation community, and (vi) Desmoulin's whorl snail.

7.3 Implications of climate change for management

Hydrological impacts of climate change have been found to vary considerably over relatively small distances within the Observatory. In the River Lambourn and Westbrook relatively small changes in channel stage and velocity are seen in response to climate change. Changes in water levels in the south meadow replicate those in the channels. These are muted compared to the north meadow, where an exaggerated response of water levels is seen especially in discrete areas of groundwater upwelling. Ecological impacts of climate change correspond with the hydrological responses. Results of this study suggest that without any modifications to current management practices, declines in the MG8 community and Desmoulins whorl snail would be exacerbated under the majority of the climate change scenarios. An overall reduction in physical habitat availability is projected for brown trout, more so for adult than juvenile life stages. Periods of low flow are particularly associated with a pronounced climate change-induced reduction of available physical habitat. Again, without modifications to current management practices, the available physical habitat for brown trout would decline under the majority of climate change scenarios.

In the wetland, aside from water levels, it is also important to consider how the degree to which water sources interact will affect plant species distribution through the available nutrient budget. Hydrochemical analysis has shown that chalk groundwater upwelling into the peat contains high concentrations of NO_3 and SO_4 and low P concentrations. Elsewhere, the peat contains reducing waters low in NO_3 and SO_4 , yet high in P. These different chemical environments have been found to promote distinct plant species, as discussed in Section 3.11.3 and corresponding to wider research (Grootjans et al., 1988; Wassen et al., 1988; Lucassen et al., 2006). High concentrations of nitrate supports localised growth of greater tussock sedge (*Carex paniculata*) within surrounding fen communities in higher phosphate waters. Lowering of water levels will promote aerobic conditions within the peat, whilst increases in groundwater contributions will cause further changes in the chemical environment. Such changes could have localised ecological effects that are not accounted for by generalised site management practices. Aerobic conditions can also lead to peat oxidation and release of CO_2 , whereas wet peat may generate methane. Soil water levels are therefore important for managing greenhouse gas emissions from wetlands (Acreman et al., 2011).

Uncertainty in the change in climate projected by the HadRM3 RCM for the 2080s converts directly into uncertainty in the hydrological and subsequent biotic response at the Observatory. Simulated water levels differ between scenarios spatially, in magnitude,

and in sign of change, exhibiting seasonality over time. Inter-ensemble variation is larger in the north meadow in comparison to the south meadow and watercourses, especially during periods of high water through the winter. In the north meadow four scenarios suggest an overall increase in mean water levels throughout the simulation period while the remaining seven show mean declines. The same directional trend is seen for summer periods of low water levels, while in winter six of the 11 scenarios result in increases in mean water levels. In the River Lambourn and south meadow the general trend of change is negative with nine out of the 11 scenarios showing lower water levels for the complete simulation period and the summer. In winter eight of the scenarios show lower levels, one no appreciable change and only two project increases.

Such uncertainty in projected changes in water levels would generally lead to a consideration of adaptive management (Walters and Hilborn, 1976; Millar et al., 2007; Lawler et al., 2008). Implementation of adaptive management is an iterative process. The outcomes of management actions are combined with system monitoring; these inform new actions to address changes in system state (Holling, 1978). This may be passive, basing strategy on historical data and adjusting as new data are gathered, or active, where the outcomes of direct experiments are assessed (Walters, 1986). The advantage of such an approach lies in the ability to cope with uncertainty through a structured improvement of relevant knowledge, whilst minimising the risks associated with on-going management (Keith et al., 2011). However, there are relatively few instances where adaptive management has been applied to practical issues (Walters, 1997; Keith et al., 2011), and these generally fall into what may be termed 'trial and error management' (Duncan and Wintle, 2008).

At the Observatory, adjustments in channel stage have profound and differing ecohydrological impacts throughout the site. The most noticeable of these impacts are a result of instream management of macrophytes through weed cutting. Ecological stresses due to changes in wetland water levels and channel hydraulics are aggravated under the majority of climate change scenarios. Yet, without weed cuts, it is debatable whether flow responses to climate change would have had such subsequent effects on habitat availability, or wetland water levels would have become intolerable. The differences in the water level requirements of the MG8 community and Desmoulin's whorl snail suggest that there is some potential to manage the wetland for the promotion of each in different parts of the site. In the River Lambourn, changes in habitat availability for brown trout are depth limited, rather than velocity limited, by the hydraulic geometry. There is potential to rethink the existing management regime for macrophytes in order to reduce negative environmental effects whilst maintaining flood resilience, yet also

mitigate the impacts of climate change. The River Lambourn is designated for brook lamprey, which are migratory. Hard engineering, regulating structures such as sluices or weirs, would obstruct the migration route from nursery beds to spawning grounds. Hence, a soft engineering approach is needed. Control of water levels through the site by modifying weed cuts could provide a multiple objective and passive adaptive management scheme able to deal with current biotic declines and potential future pressures associated with climate change.

7.4 Recommendations for multiple objective management

A reassessment of the existing management regime is required to target different ecological requirements in different areas of the Observatory. Key species and communities include those for which the site is designated, i.e. Desmoulin's whorl snail, the MG8 plant community and brook lamprey, plus those which are productive, namely brown trout. Efforts may be split spatially between the north and south of the site, with a reduction in the frequency/severity of weed cutting in the north and adjustments to the schedule of this management in the south (Figure 7-4). These actions are capable of providing a range of benefits through the river and wetland, and may be adjusted to account for the effects of climate change.

7.1.1. Management actions in the north of the Lambourn Observatory

In the north meadow, survival of Desmoulin's whorl snail depends on the maintenance of high water levels and standing water (Section 6.2.1). Upwelling Chalk groundwater locally elevates water levels and provides potential areas of refuge. However, water levels rarely reach desirable elevations even in these areas. The more extreme of the climate change scenarios marginally lengthen the time over which such levels persist, while projections associated with lower water levels result in periods of intolerable hydrological conditions. A reduction in the severity of weed cuts in channels bordering the north meadow would augment and maintain the high water levels necessary for Desmoulin's whorl snail (Figure 7-5). To help the tall-herb fen habitat persist no meadow vegetation would be removed through cutting or grazing.

Modifications to the weed cut may include a decrease in frequency, restricting cutting to once a year, or modifying the weed cut pattern to remove less (below 10-20% for instance) yet still maintain sinuosity of flow. In the channel, the subsequent maintenance of high stage would increase the available physical habitat for brown trout in both juvenile and adult life stages, which are most vulnerable to climate change during periods of low water level. A relative increase in the amount of vegetation would also benefit fish

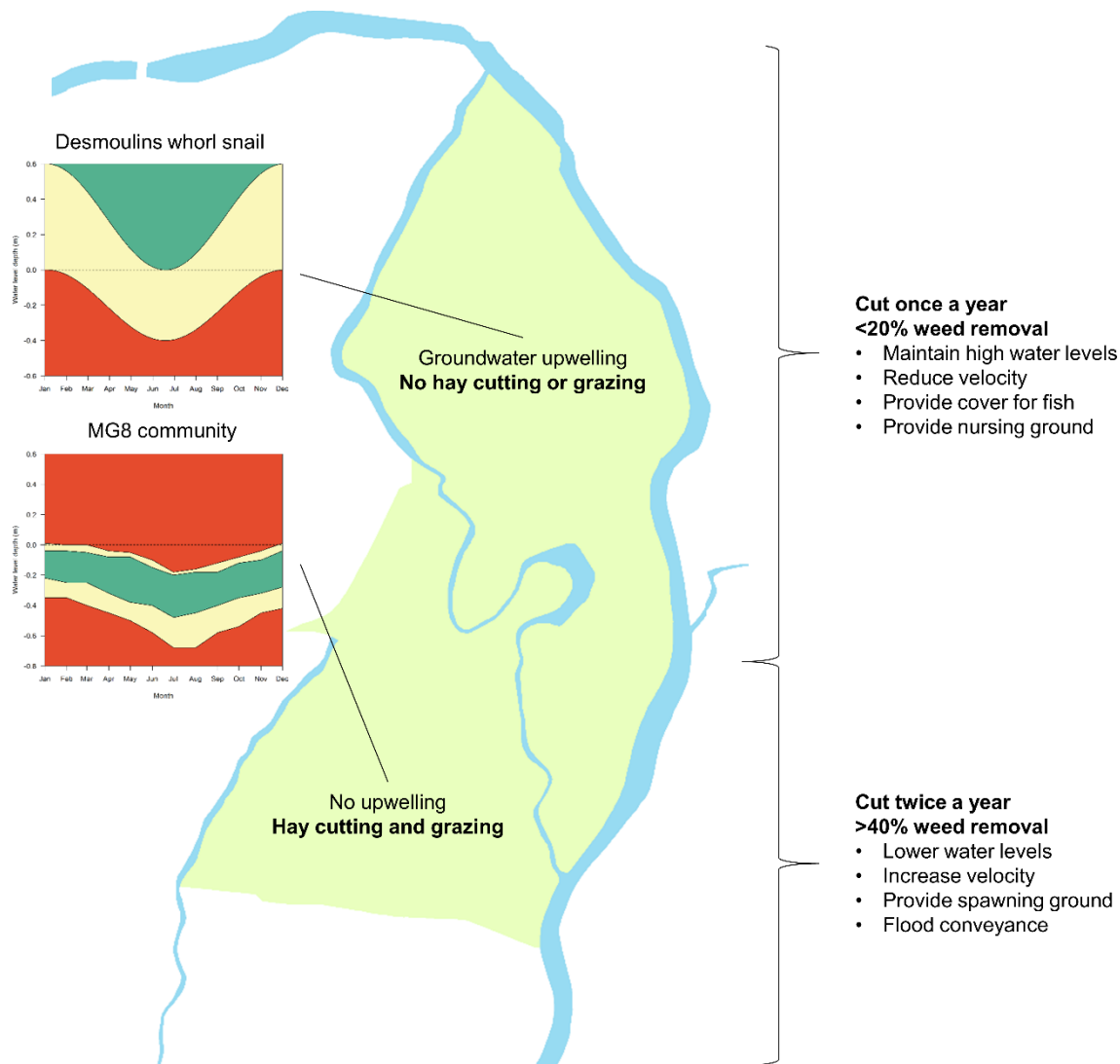


Figure 7-4. Map showing recommended management actions and their potential effects in different regions of the Observatory

species by providing more habitat for invertebrate prey, cover for protection, and oxygen production (Bursche, 1971). Reduced velocities and organic detritus may lead to siltation of the bed substrate and a reduction in the area of clean gravels necessary for spawning. However, the preferred substrate for juvenile brook lamprey has a relatively high organic content, and is composed of mud, silt, or silt and sand (Hardisty and Potter, 1971; Maitland, 2003). The soft sediment provides habitat and food (Clemens et al., 2010). Juvenile trout survival and abundance in chalk streams have been shown to positively correlate to flow in the preceding April (Solomon and Paterson, 1980). The stages and velocities sustained through spring macrophyte growth are thus critical to trout production. A section of reach with a relatively high volume of macrophytes would provide a nursing ground and shelter for both brown trout and brook lamprey.

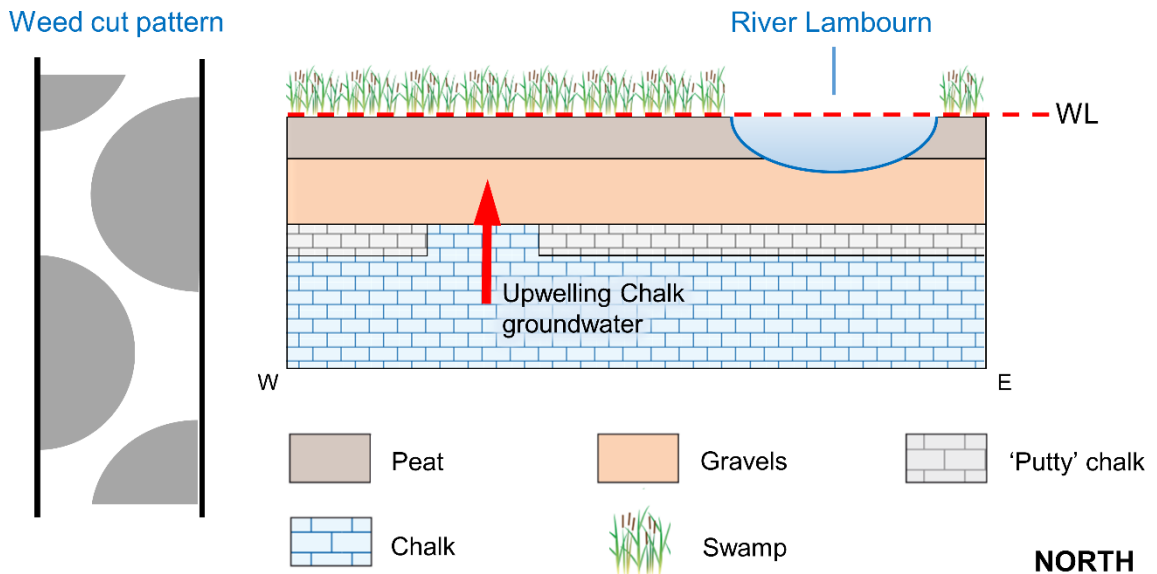


Figure 7-5. Proposed weed cutting pattern for the north of the Observatory and schematic showing resultant water levels in the north meadow

7.1.2. Management actions in the south of the Lambourn Observatory

In the south meadow, without groundwater contributions from the Chalk aquifer, water levels are principally controlled by channel stage. These water levels, although generally within the requirements for the MG8 vegetation community, frequently stray into dry intolerable conditions. Summer periods of low water level aggravate the impacts of preceding weed cuts, and are also the time of greatest vulnerability to climate change impacts. Continuation of >40% weed removal in channels adjacent to the south meadow would help keep conditions desirable in periods of high water level (Figure 7-6). Adjusting the timing of weed cuts to fit the seasonal water level requirements for MG8 would help maintain the levels during periods of high water level that are needed for its development. Dependent on prevalent water levels, seasonal or climate change-induced, there would be some adaptive change in the severity of the cuts, though this would be a subsidiary action.

To sustain the MG8 community, weed cuts should be complimented in the south meadow by a schedule of mowing and grazing (Wheeler et al., 2004). This form of vegetation management prevents tall species such as sedges (*Carex* spp.), sweet-grass (*Glyceria* spp.) and rushes (*Juncus* spp.) dominating. A typical schedule involves a midsummer hay cut followed by autumn grazing of re-growth. Hay cut and removal may be important for the removal of nutrients, which would otherwise accumulate in the system.

Removal of a relatively high proportion of the weed mass (>40%) in the channels would provide the change in hydraulics needed for other objectives. The preferences of both

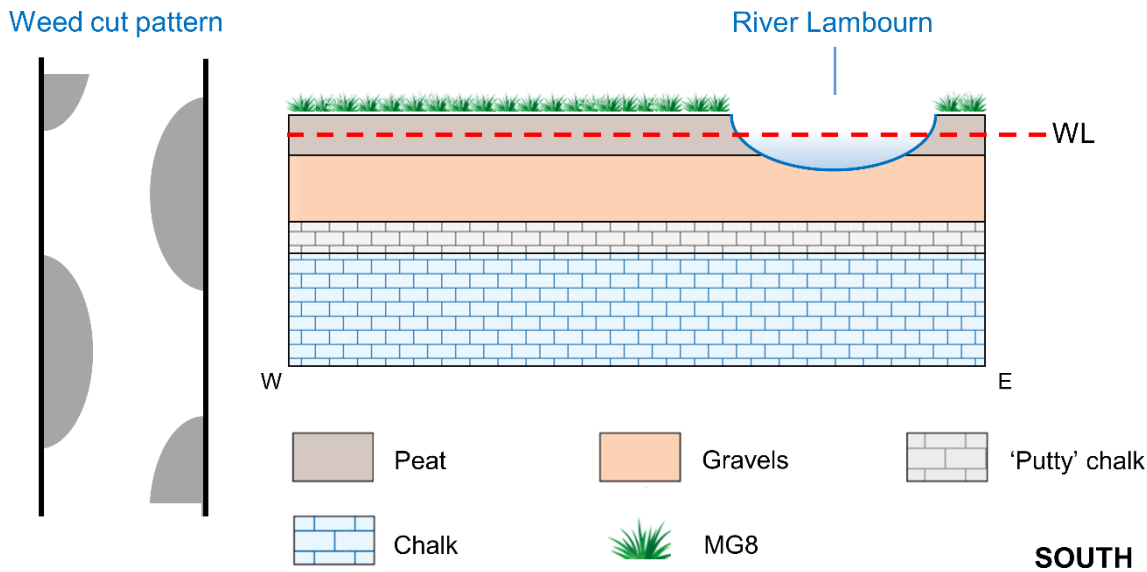


Figure 7-6. Proposed weed cutting pattern for the south of the Observatory and schematic showing resultant water levels in the south meadow

brook lamprey and trout for spawning beds are likely to be similar (Maitland, 2003). Productive spawning beds are composed of permeable gravels with a proportion of smaller material for stabilisation (Stuart, 1953). These meet the requirements of eggs for oxygen and shelter. Fine sands or silts are unfavourable as they harden the bed and lead to impermeability. The higher velocities which result after weed cutting mobilise sediment and ensure that clean gravel spawning habitats are maintained (Soulsby et al., 2001). Removal of vegetation to this specification reduces flow resistance and stage, increasing the conveyance capacity of the river. This mitigates local flood risk and would also help mitigate the possible increases in flow due to climate change.

7.1.3. Considerations for effective management

Any adjustments to the management regime would be dependent on seasonal flow and water levels combined with any changes induced by climate change. These would be necessarily adaptive and require continuous monitoring. Constraints on resources call for a minimal monitoring network requiring little maintenance. The existing network could be reduced to a few indicative piezometers and stage boards, yet still meet management needs (Figure 7-7). Although, any reduction in the monitoring programme or network could impact the potential for future scientific study. As the proposed recommendations are a modification of practices already carried out at the Observatory they would be inexpensive and straightforward to implement.

The mitigation of flood risk through reductions in stage and increases in conveyance capacity are a key feature of the weed cuts. The reduced flow resistance and volume

decrease from vegetation removal increases the conveyance capacity of the river (Old et al., 2014) and reduces stage, so mitigating local flood risk. This has important social and economic implications for the surrounding community, which previous events have also shown to be politically charged (e.g. Pitt, 2008; Morris and Brewin, 2014). However, this study has indicated largely detrimental ecological impacts, offset by a reduction in flood risk, a sensitively balanced issue with wider implications. There is a delicate balance between managing for flood conveyance and the conservation of desired species, whilst also incorporating measures to account for potential effects of climate change. Additional sociopolitical aspects for the surrounding community in terms of flood resilience and natural capital would need to be factored into any management scenario.

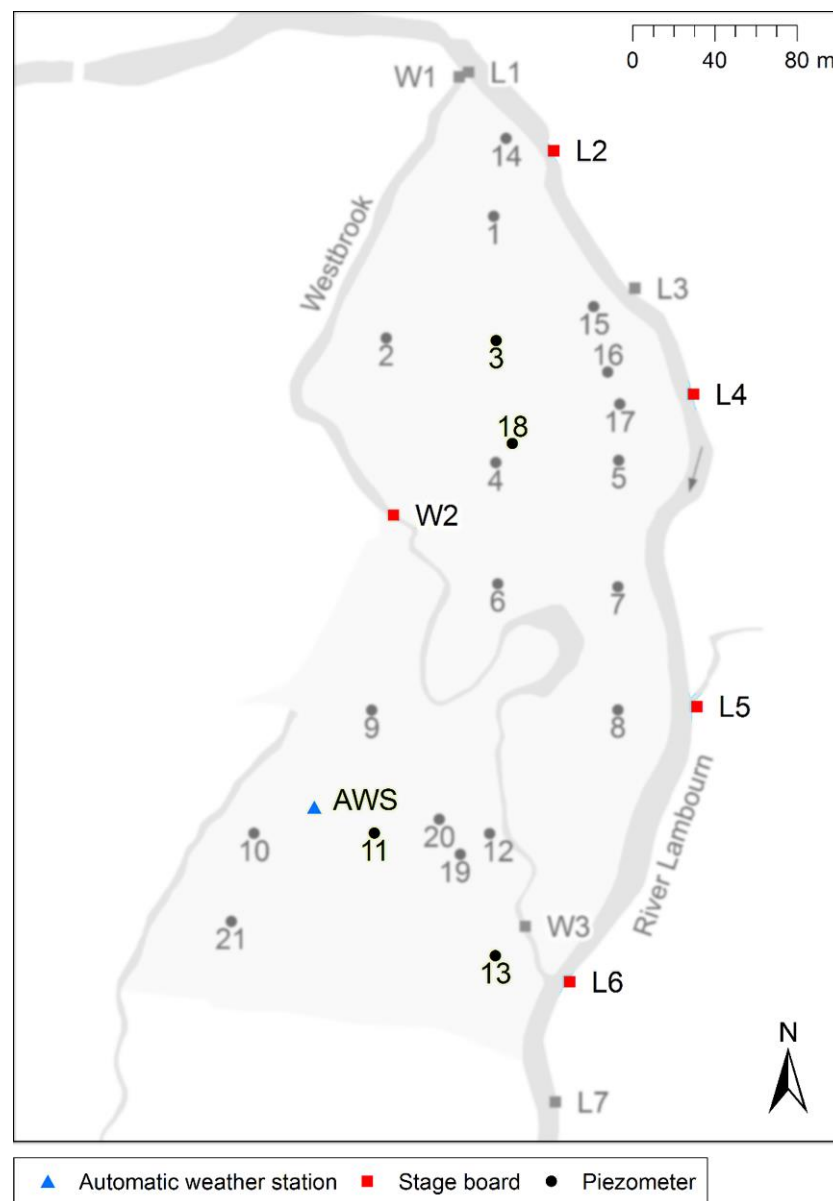


Figure 7-7. Proposed reduced extent of the instrumentation network. Locations are highlighted with the existing network greyed out

The climate change scenarios employed in this study were derived from the Future Flows and Groundwater Levels project (Jackson et al., 2011; Prudhomme et al., 2012) (Section 5.2). This was based on HadRM3-PPE-UK, containing a set of transient climate projections used in derivation of the UKCP09 scenarios (Murphy et al., 2007). HadRM3-PPE-UK was designed to simulate UK regional climate for the period 1950–2100 for historical and SRES A1B emissions scenarios (Murphy et al., 2009). However, while it represents parameter uncertainty through a parameter variant of climate sensitivity, the ensemble under samples GCM uncertainty and excludes emissions scenario uncertainty. To incorporate different emissions scenarios, UKCP09 also provides probabilistic projections under three emissions scenarios, Low, Medium and High, which correspond to the SRES B1, A1B and A1FI Scenarios (Jenkins et al., 2009). Projections for atmospheric variables take the form of a probability distribution function, intended to represent the uncertainty in future climate. A range of probabilities around the 50% probability central estimate of change, as likely as not to be exceeded, may be taken to incorporate different emissions and severities of climate change. Although this would add further uncertainty to any management decisions, it would help constrain the expected extent and likelihood of any climate change-induced changes.

Uncertainty in the ecohydrological impacts of modifications to the weed cutting regime means that suggestions for precise management actions are questionable. The MIKE SHE / MIKE 11 and physical habitat modelling system employed in this study could be used to constrain some uncertainty by simulating the impacts of weed cutting strategies with different severities, spatial application and timing. The impacts of a range of weed cut options upon wetland water levels and available physical habitat could be assessed under both current and scenario climates. These could be used to define a range of selected management responses for different scenarios, creating a go to resource to aid conservation managers in adapting to system changes.

7.5 Summary

Without any modifications to current management practices within the CEH Lambourn Observatory, declines in the MG8 community and Desmoulins whorl snail would be exacerbated under the majority of the climate change scenarios. Climate change is also projected to induce an overall reduction in physical habitat availability for brown trout, more so for adult than juvenile life stages. However, the current management practice of cutting instream vegetation itself has largely detrimental ecological impact. Conditions deteriorate following weed cuts for species and communities for which the site is designated, namely Desmoulin's whorl snail, the MG8 plant community and brook

lamprey, and also the productive species of brown trout. These ecological impairments are offset by a reduction in local flood risk, which is a politically sensitive issue with wider implications.

Different hydrological requirements for species in distinct areas of the site indicate multiple objective management is possible. This may be achieved with modifications to the existing management regime. It is recommended that actions be divided between the north and south of the site. A reduction in the frequency/severity of weed cutting in the north will maintain the high wetland water levels needed for the survival of Desmoulin's whorl snail, increase habitat availability for brown trout, and provide a nursing ground for brook lamprey and brown trout. In the south, adjustments in the cutting schedule with a continuation in the severity of weed cuts would maintain desirable wetland water levels for the MG8 vegetation community, provide spawning grounds for brown trout and brook lamprey, and mitigate for flood risk. The proposed management scheme would be adaptive to account for the projected uncertainty in the impacts of climate change. Continuous monitoring would be needed though this could potentially be diminished. However, though reduced monitoring may satisfy management requirements it could have opportunity costs in terms of future science. The proposed recommendations for management are economical and can be implemented easily.

Chapter 8

Conclusion and Recommendations for Further Research

8.1 Introduction

Ecohydrological response to climate change was explored at the CEH River Lambourn Observatory, Boxford, UK. The site comprises a 10 ha chalk valley bottom, riparian wetland and 600 m reach of the River Lambourn designated for its conservation value and scientific interest. Such valley bottom wetlands of chalk catchments are discontinuous but widespread. Here, the physically based, distributed model MIKE SHE was used to simulate the hydrology of the Lambourn Observatory. Building on a robust conceptual model, the hydrological impacts of climate change were assessed for the river and wetland site. These were linked to water level requirements for the MG8 plant community and Desmoulin's whorl snail in the wetland, and habitat availability for brown trout in the river. Impacts on each are shown to differ spatially, with implications for different management strategies across separate areas of the study area.

Concluding the thesis, this chapter revisits the aim and objectives, providing an assessment of how completely they have been achieved. Limitations of the study and recommendations for further research are discussed, as are key findings that have wider implications for the region and will be of interest to both the research community and conservation organisations.

8.2 Assessment of aims and objectives

The aim of this study was to assess the ecohydrological impacts of climate change on a riparian wetland in the chalk lowlands of the UK, with particular reference to the CEH River Lambourn Observatory. This has been accomplished through the completion of five objectives:

1. *Develop a conceptual model of the wetland through synthesis of existing research and further data acquisition*

A review of existing information on the site and its catchment (Chapter 2) and results from a fieldwork campaign targeted to address knowledge gaps (Chapter 3) led to the development of a conceptual model of hydrological functioning for the Observatory. In general, topographical slope, along with head contours in the peat, gravels and chalk, follow the line of the Lambourn valley. A network of shallow, infilled relic channels crosses the site connecting to the channels. Flows in these channels display a non-flashy

regime with continuous baseflow and runoff dominated floods. In-channel macrophyte growth and its management dominate the fluvial regime and have an important control on water levels. The peat and gravels are considered to have good hydraulic connectivity, with head boundaries at the River Lambourn and Westbrook broadly controlling water levels across the wetland. A double aquifer system of gravels and Chalk is mostly separated by a confining layer of low permeability 'putty' chalk. Leakage occurs between the gravels and Chalk where the putty chalk is thin or absent, causing localised variations in water levels. These are concomitant with the relic channels in the peat and occur mainly in the north meadow.

Although the conceptual model is considered robust, and validated by the MIKE SHE model (Chapter 4), spatial surveys were restricted in coverage of the site. Surveys reliant on GPS were limited in area due to poor accessibility and connectivity to the satellite network in areas of high vegetation. For topography, this shortcoming was addressed through the use of LiDAR data. However, for the geophysical and temperature surveys, features beyond the survey extents may have been missed. To illustrate, a temperature anomaly and area of persistently high groundwater head elevation at the northern tip of the site was assumed to be due to chalk groundwater upwelling through a gap in the putty chalk (Sections 3.6.2, 3.10 and 4.3). As this was outside the geophysical survey extent the validity of this assumption is uncertain, and hints at the possibility of other anomalies within the site that would affect its hydrological functioning. The temperature survey was, additionally, limited to a snapshot in time. Seasonality of groundwater discharge or dynamic response to intense precipitation events would not be captured in a single temperature survey, while the rate of groundwater discharge was unquantifiable.

2. Develop a distributed hydrological/hydraulic model to simulate wetland hydrology and to produce a water balance of the site

The physically based, distributed model MIKE SHE was chosen to simulate hydrological processes at the Observatory (Chapter 4). An ability to adapt the complexity of process representation to the applications, a modular framework, and the calculation of each process at distinct and appropriate time steps and spatial scales distinguished MIKE SHE from other relevant modelling systems. Model results were generally consistent with field observations, confirming the conceptual model. A water balance showed hydrological processes in the wetland to be dominated by the interaction between groundwater and surface water. Channel stage provides head boundaries for broad water levels across the wetland, whilst areas of groundwater upwelling control discrete head elevations. A relic surface drainage network confines flooding extents and routes

seepage to the main channels. Instream vegetation was found to affect water levels throughout the site and the interaction between surface water and groundwater. Macrophytes influence head variations across the wetland, the proportional contribution of each water source, and the maintenance of areas of standing water.

An intrinsic difficulty in representing compressible, anisotropic soils limited otherwise excellent model performance in some areas. This in particular led to inferior model performance in the peat. The incorporation of the effects of soil deformation is not limited to MIKE SHE, being generally overlooked in other hydrological modelling systems. Overestimations of head in the MIKE SHE model when water elevations are high may be explained by peat liquefaction with saturation and the associated increase in hydraulic conductivity. In addition, small-scale variations in substrate characteristics, difficult to establish in the field, could contribute to local variations in hydraulic conductivity. At the model grid resolution that was employed in the Boxford model these could have a significant impact on simulated groundwater flow and levels, and account for poorer performance in certain areas. Issues with the installation of peat piezometers from the pre-existing array, where the slotted screen extending above ground level could cause influx of surface water. This issue could have produced misleading results and impacted model performance. However, this has not been directly observed and poor performance in some areas is more likely a result of substrate variations and peat compressibility.

In the channels, localised effects of macrophyte growth on bed roughness, not included within the model, may account for discrepancies between simulated and observed channel stage. An unmonitored sluice gate located in the Westbrook could additionally account for poor representation of channel stage in certain areas and at certain times, whilst the cause of relatively inconsistent heads around the Westbrook is also unclear. Although the area falls beyond the extents of the detailed topographical and geological surveys, it is difficult to assess whether results are from a real or interpolated feature, and highlight the importance of high-resolution field data.

3. Project changes in hydrometeorological inputs to the hydrological/hydraulic model under scenarios of different climate sensitivities to incorporate the uncertainty associated with climate change

Projected changes in precipitation, potential evapotranspiration, channel discharge, and groundwater level for the 2080s were derived from the UK Climate Projections 2009 using datasets from the Future Flows and Groundwater Project (Chapter 5). Although emissions scenario uncertainty was excluded, parameter uncertainty was represented through variants of climate sensitivity. A delta factor approach was used to obtain

monthly percentage differences between the ensemble reference period (1961–1990) and the future period (2071–2098), which could be used to perturb inputs to the MIKE SHE model. Although this approach assumes that climate variability does not alter and provides no information on changes in event frequency and distribution, it enables a robust comparison of average outcomes and has been widely used in hydrological studies of climate change (e.g. Arnell and Reynard, 1996; Limbrick et al., 2000; Kamga, 2001; Arnell, 2004; Thompson et al., 2009; Jackson et al., 2011). Drier summer and wetter winter months were evident from the precipitation and PET change factors. However, the magnitude, direction and duration of changes varied considerably between scenarios, reflecting the uncertainty linked to climate change. Additionally, a general decline in river flow and increase in Chalk groundwater levels was projected. Although, at first, this may seem counterintuitive, the gravel and Chalk aquifers may experience differing responses to climate change, with the effect that the mostly gravel aquifer influenced river will display reductions in discharge, whilst the mostly separated Chalk aquifer will show increases in head. However, the projections may be a result of inadequacies in resolution and boundary conditions in the regional groundwater model and/or rainfall-runoff model.

4. Use the model to investigate how climate change scenarios affect wetland hydrology

Projected changes in hydrometeorological drivers were applied as inputs to the MIKE SHE distributed hydrological/hydraulic model of the Lambourn Observatory (Chapter 5). Although the simulation period was relatively short, 01 Feb 2013 – 01 Oct 2014, due to data availability, the approach replicates those used elsewhere (e.g. Thompson et al., 2009) and used fine resolution data. The simulated hydrological impacts of climate change were found to vary considerably over relatively small distances due to differences in groundwater/surface water interaction and water availability. Climate induced changes to the hydrological functioning of the Observatory reflect the site's position in a chalk valley bottom. Water supply mechanisms are influenced by meteorological changes occurring over the catchment and regional area.

It was not possible to drive the entire chalk boundary of the MIKE SHE model of the Lambourn Observatory with predictions from the regional groundwater model as the grid was too coarse. This meant a pressure boundary was employed around the entire model, with the subsequent issue that the water balance was not easily verified as it would potentially be sensitive to hydraulic conductivity in the Chalk. The model was found to be sensitive to vertical hydraulic conductivity in the Chalk, although not horizontal hydraulic conductivity, which was employed as a manual calibration parameter in the baseline

model since it predominantly affected bias. A further disadvantage of a hydrological model at the site scale lies in the ability of the boundary conditions to represent flow changes from the wider area. Regional changes in precipitation and evapotranspiration would be expected to translate to comparable changes across the flow boundaries. The comparison of total boundary inflow under each scenario to precipitation showed that changes are within the same magnitude. Additionally, the good relationship between total boundary inflow, precipitation and evapotranspiration indicated that the effects of climate change are accounted for by the modelling approach used.

5. *Compare simulated hydrology under each climate change scenario with the requirements of species / communities for which the site is managed to assess ecohydrological impacts of climate change and resulting management implications*

Simulated wetland water levels were linked to requirements of the MG8 plant community and Desmoulin's whorl snail (*Vertigo moulinsiana*) for which the site is designated (Chapter 6). Impacts on each were shown to differ spatially and in line with hydrological impacts. In the river, assessment of river habitat response of brown trout (*Salmo trutta*) to climate change employed a modified PHABSIM habitat modelling methodology using outputs from the 1D hydraulic module of MIKE SHE, MIKE 11. Impacts of climate change were seen as most pronounced through summer months, and accentuated periods of low flows. These were reflected in a marked reduction in habitat availability, more so for adult than juvenile trout. Differences in hydrological requirements for species in distinct areas of the site support separate management strategies (Chapter 7). Recommendations for management were apportioned spatially. In the north of the site, a reduction in the frequency/severity of weed cutting would maintain the high wetland water levels needed for the survival of Desmoulin's whorl snail, increase habitat availability for brown trout, and provide a nursing ground for brook lamprey and brown trout. In the south, a schedule adjustment with a continuation in the severity of weed cuts would maintain desirable wetland water levels for the MG8 vegetation community, provide spawning grounds for brown trout and brook lamprey, and mitigate for flood risk. Multiple objective management may be achieved through adaptive modification of the current management regime.

It is realised that the water level requirements provided by the literature indicate the broad range of hydrological regime that gives rise to specific vegetation communities. Nevertheless, for the MG8 community highlighted in this study there are detailed data to identify the magnitude of hydrological impact that would have an effect. Hydrological requirements for Desmoulin's whorl snail are uncertain, but did provide some indicative

water levels that were able to be used to assess the potential impacts of climate change on this individual species. Habitat Suitability Indices for brown trout are also ambiguous, derived from observations of fish placement in the field and expert opinion. Nevertheless, requirements represent the best available information and are considered robust. Additionally, climate change will have implications for other environmental factors influencing wetland and riverine ecosystems, such as nutrient supply, that are not addressed in the current study. Direct effects of climate change on vegetation through increases in temperature and carbon dioxide have, in addition, not been accounted for. Moreover, the water requirements of particular species and communities may themselves change with the climate, a factor also beyond the scope of this study, yet which could have an influence on their maintenance or succession.

8.3 Key research findings

Further to the completion of the aim and objectives, a number of key findings may be drawn from this study that have wider implications and significance for wetland hydrology and hydroecology:

1 Existence of the wetland is due to an impermeable layer of 'putty' chalk

Valley bottom wetlands within chalk catchments are discontinuous but widespread. However, on permeable geology and with the relatively low levels of annual precipitation for southeast UK, the reason for their existence has been unclear. A layer of low permeability 'putty' chalk is present at the interface between the Chalk and alluvial gravel aquifers. Although it is discontinuous, the putty chalk has been shown to act as a largely confining layer to the Chalk aquifer, restricting flow exchange. This impermeable layer, along with the overlying gravel aquifer, would account for the stable water level regime necessary for the maintenance of the wetland and its counterparts throughout the region.

2 Groundwater/surface-water interaction dominates hydrological processes

Surface water and groundwater are inextricably linked and crucial to processes in the wetland. The water balance developed with the MIKE SHE model indicated proportional contributions of 44.2% for surface water, 43.4% for groundwater, 6.3% for precipitation, and 5.7% for actual evapotranspiration. The channels, gravel aquifer and peat are hydraulically connected. Channel stage acts as a head boundary and controls broad water levels within both the gravels and peats. Responses in groundwater flow from the Chalk aquifer are also head controlled by the channels. Increased stage resulting from storm events raises head elevations within the gravels and peats, inhibiting upwelling from the chalk groundwater. Conversely, reductions in head from weed cuts in the River

Lambourn draw water up from the chalk, increasing the rate of upward groundwater flow. The influx of surface water into the gravels at high stage drives the increases in gravel head. Groundwater/surface-water interaction defines the hydrological functioning of the wetland, which is likely to be reciprocated in similar wetlands across the region.

3 Regional and catchment processes control water supply mechanisms

Hydrology at the Observatory reflects larger-scale catchment processes and the position of the site in a chalk valley bottom with a groundwater fed river. The longer-term trend of surface water outflow follows the seasonal pattern of groundwater inflow. When heads in the Chalk are high, so are levels in all components of the system. Climate-change induced seasonal modifications to scenario precipitation and year-round increasing potential evapotranspiration cause wetter winters and drier summers. Although these have some direct influence, changes to wetland water levels are mostly governed by the projected changes in discharge and groundwater level. Hydrometeorological projections indicate declining river flow and increasing groundwater levels for the site. Although this seems counterintuitive, the gravel aquifer accounts for a down-valley component of groundwater flow through the Lambourn catchment. Water from the gravel aquifer has been shown to provide a significant contribution to the site. Yet, whilst the gravels are in good hydraulic connectivity with the river, hydraulic connection to the Chalk is variable. Thus, differing responses to climate change in each aquifer are possible. The mostly gravel aquifer influenced river will display reductions in discharge, whilst the mostly separated Chalk aquifer will show increases in head. These in turn are influenced by meteorological changes occurring over the catchment and regional area.

4 Discrete areas of groundwater upwelling cause local variations in water levels

Discrete areas of groundwater discharge were found in areas that had not previously been considered to be groundwater dependent. These were delineated with a high spatial resolution 3D temperature model, after being overlooked with point measurements of groundwater head. Groundwater from the Chalk aquifer is an important source of water into the Observatory, discharging into the gravel aquifer and wetland through gaps in the putty chalk and resulting in locally varied heads. These are concurrent with relic infilled channels in the peat and occur in the north meadow. Such features are likely to be widespread within riparian lowland wetlands and could be pivotal in the hydrological functioning of many sites, yet the conceptualisation may lack this information. A temperature model may thus be an effective precursor to the targeted installation of more costly monitoring, in this study an array of piezometers. Such targeted piezometer arrays should be favoured in these heterogeneous valley bottom

settings where data from dense random clusters or gridded arrays have been demonstrated to be insufficient for successful site characterisation.

5 Groundwater supports a distinct plant species, useful as a botanical indicator

Site conceptualisation revealed the prevalence of *Carex paniculata* (greater tussock sedge) in areas of deeper groundwater upwelling suggesting that it could be a useful botanical indicator of groundwater dependence. This correlation is considered to be a result of the relatively NO₃ and SO₄ rich, Ca and P poor, waters associated with the deeper groundwater, as opposed to a causal link to temperature. *Carex paniculata* is likely limited by N rather than P. Surrounding reducing waters low in NO₃ and SO₄ and high in P promote poor fen communities. A need for hydrochemical or hydrogeological supporting information limits the use of *Carex paniculata* as an indicator species to preliminary site assessments. Nevertheless, in similar geological settings the species may help determine further investigative requirements.

6 Ecohydrological conditions in the wetland differ over short distances

Wetland ecology differs spatially and in line with variations in water levels. Localised areas of upwelling chalk groundwater cause elevated water levels in contrast to areas controlled by heads in the river and alluvial gravel aquifer. Hence, differences in water level requirements for protected species/communities, in this study the MG8 plant community and Desmoulin's whorl snail (*Vertigo moulinsiana*), contribute to their survival in distinct areas. The designated biota is currently in decline and climate change is projected to deteriorate conditions under most scenarios. In the groundwater influenced north of the site Desmoulin's whorl snail is most likely to survive where the high water levels and standing water necessary for its survival are likely to continue. In the south where channel stage controls water levels the MG8 community has a much better chance of recovery. Changes in ecology due to differences in groundwater/surface-water interaction and water availability have important and wider implications for the management of conservation priority species.

7 Instream vegetation influences river hydraulics and in turn wetland hydrology

Instream vegetation growth and its management through cutting have acute effects on water levels and velocity. Seasonal growth patterns and the subsequent increases and decreases in vegetative volume are reflected by the gradual rise and fall of channel stage. Weed cuts are associated with an abrupt drop in stage and increase in velocity. As the channels act as head boundaries to the wetland, hydraulically connected to the

gravels and peat, water levels throughout the site respond similarly. The interaction between surface water and groundwater is also markedly affected by weed cutting. As noted above, responses in groundwater flow from the Chalk aquifer are also head controlled. Thus, macrophytes influence head variations across the wetland, the proportional contribution of each water source, and the maintenance of areas of standing water. As macrophytes are features of the chalk streams of southern and eastern UK, this has implications for the extensive network of riparian wetlands that exist on their floodplains.

8 Impacts of climate change are uncertain and vary spatially over a small scale

Uncertainty in the change in climate projected by the RCM for the 2080s converts directly into uncertainty in the hydrological and subsequent biotic response at the Observatory. Simulated wetland water levels and channel hydraulics differ between scenarios spatially, in magnitude, and in sign of change, exhibiting seasonality over time. Variation between scenarios is larger in the north meadow in comparison to the south meadow and watercourses, especially during winter periods of high water level. The impact of climate change, and other environmental changes such as groundwater abstraction, will have differing responses in terms of water availability dependent on the position and size of the wetland within the catchment. Within such wetlands, due to complex hydrological relationships between bedrock and alluvial aquifers, as well as river stage, and also the interplay between site scale and catchment processes, responses may vary dramatically over relatively small spatial scales.

9 Adaptive channel management achieves multiple objectives

Different hydrological requirements for species in distinct areas of the site highlight the potential for multiple objective management. Control of water levels through modification of the existing management regime could resolve current biotic declines, mitigate flood risk, and limit impacts of potential changes in climate. Targeted spatially, an adaptive management regime would be needed to account for seasonal changes in flow and water levels, along with the effects of any environmental changes induced by climate or other factors such as abstraction. A reduction in the frequency/severity of weed cutting in the north will maintain the high wetland water levels needed for the survival of Desmoulin's whorl snail, increase habitat availability for brown trout, and provide a nursing ground for brook lamprey and brown trout. In the south, a schedule adjustment with a continuation in the severity of weed cuts would maintain desirable wetland water levels for the MG8 vegetation community, provide spawning grounds for brown trout and brook lamprey, and mitigate for flood risk. Constituting adjustments to the management regime, such

recommendations for site management are economical, easy to carry out, and could be transferred to similar settings.

10 Physically-based integrated models such as MIKE SHE are effective at representing and developing system understanding at high spatial resolution

The potential of physically-based models, in this case MIKE SHE, to model wetlands with complex subsurface architecture has been demonstrated for the River Lambourn Observatory. Numerical model results supported the conceptual model, with generally very good to excellent model performance. Results were consistent with field observations and followed short-term responses to hydrological and management events, representing the impact of instream weed cutting well, as well as the longer-term seasonal cycle. Development of a wetland hydrological/hydraulic model allowed quantification of the water balance and enhanced understanding of the site's hydrological functioning. In the wetland, water levels were simulated at the high resolution needed to link water table results directly to plant and animal requirements, allowing the assessment of ecological impacts of climate change. In the river, application of the MIKE 11 hydraulic component of the MIKE SHE model has elucidated the variability in flow conditions and, thus, habitat availability for a macrophyte dominated river. The use of outputs from hydraulic models in assessing impacts of environmental change on physical habitat availability is cost-effective, efficient, and has international applicability. The MIKE SHE model has shown potential for investigating the hydrological effects of environmental changes, in this case from climate change. It could be readily employed to assess changes due to groundwater abstraction as well as changes in channel morphology or vegetation management. Physically based, integrated models, such as the MIKE SHE model of the Lambourn Observatory, therefore represent effective and valuable tools for understanding wetland ecosystems and their response to change and for developing management approaches.

8.4 Recommendations for further research

In this thesis, a fieldwork campaign and hydrological/hydraulic model have illuminated wetland hydrological processes and groundwater/surface-water interaction, which, when linked to ecological requirements, allowed assessment of climate change impacts and recommendations for conservation management. A number of areas may be identified where it is possible to build on and further improve understanding:

a) Quantification of groundwater flux

Estimates of the rate of groundwater flux in upwelling areas would help verify the water balance and provide additional observations to constrain the calibration parameters. One-dimensional vertical temperature arrays are becoming increasingly commonplace as a means of estimating groundwater fluxes (Anibas et al., 2009; Voytek et al., 2013). Such estimates require profile time series to solve analytical flux representations (Hatch et al., 2006; Briggs et al., 2012). The detailed 3D temperature model could be a useful precursor to the targeted deployment of sensor arrays. Sensors could be positioned to sample temperature gradients representatively across an entire site to estimate total groundwater influx. Furthermore, their deployment could also be based upon an understanding of the flow field, which is important to avoid misinterpretation of temperature time series (Cuthbert and Mackay, 2013). For example, at the Lambourn Observatory there is evidence for non-vertical flows which have been considered the greatest source for error when implementing 1D solutions (Lautz, 2010).

Data has typically been collected in the field through use of thermocouples positioned at low spatial resolution (Lapham, 1989; Hatch et al., 2006; Keery et al., 2007). This can hinder interpretation of complex systems where high resolution data are needed. Furthermore, the optimal spacing of temperature sensors will vary with flux magnitude, which itself may vary over time and depth (Briggs et al., 2012). A high resolution of temperature measurements is crucial and could be achieved with Distributed Temperature Sensing (DTS) systems, which enable temperature measurements along a fibre optic cable buried within the wetland (Selker et al., 2006; Vogt et al., 2010; Briggs et al., 2012). Fibre optic DTS may be coiled to increase resolution and enable vertical placement, or laid level to investigate lateral flow. This would be particularly useful in the relic channels, where observed heat transport may encompass an element of lateral advection induced by flow.

b) Development of a model for flow through compressible soils

The inability of MIKE SHE, and many other hydrological models, to represent compressible, anisotropic soils has been identified as compromising model performance in the peat. Hydrological models generally apply rigid soil theory, defining the hydraulic properties of each geological unit as being temporally and spatially constant. The effectiveness of this approach has been questioned (Brown and Ingram, 1988; Baird and Gaffney, 1994), with peat saturated hydraulic conductivity being found to be highly correlated to water table depth (Price, 2003). Camporese et al. (2006) developed a model describing the variation of porosity with moisture content, valid for both anisotropic

and isotropic three-dimensional peat deformations. However, the model is not widely available and has not been validated in unconstrained field conditions. It is clear that studies of wetland hydrology, which often incorporate hydrological models, would benefit greatly from a model of flow through compressible soils that is easily accessible and may be configured to exchange data with other software components.

c) Extension of the water balance studies

To inform management decisions, the water balance could be extracted for different areas, such as the north or south meadow, and under each climate change scenario. Nutrient loadings may also be added to the water balance, which may then be assessed in the context of other information, such as the amount of nutrients that could be removed during vegetation cuts, or the response of instream nutrient concentration to changes in land management. The degree to which water sources interact will affect plant species distribution through the available nutrient budget (Wheeler and Shaw, 1995). Species response to changes in groundwater/surface-water interaction could be assessed for projected and spatially distinct water balances.

d) Probabilistic climate change scenarios

Although potentially adding further uncertainty, probabilistic climate change scenarios could help better inform management decisions by constraining the expected extent and likelihood of any climate change-induced changes. The climate change scenarios used in this study were based on HadRM3-PPE-UK, which contains a set of transient climate projections used in derivation of the UKCP09 scenarios (Murphy et al., 2007). While parameter uncertainty is represented through a parameter variant of climate sensitivity, GCM uncertainty is under sampled and emissions scenario uncertainty excluded. To incorporate different emissions scenarios, UKCP09 also provides probabilistic projections under three emissions scenarios, Low, Medium and High, which correspond to the SRES B1, A1B and A1FI Scenarios (Jenkins et al., 2009). Projections for atmospheric variables take the form of a probability distribution function. The hydrological/hydraulic model could be driven under different emissions scenarios at a range of probabilities to further incorporate the uncertainty in future climate (e.g. Thompson, 2012).

e) Extension of water level requirements to other species and communities

This study has focussed on the impacts of climate change to particular species and communities for which the Observatory is designated and managed. However, the site

is also designated for brook lamprey that are not included directly in this study, in addition to containing several non-designated vegetation and animal communities and species. HSI for brook lamprey may be available, allowing assessment of the habitat sensitivity of the species. Care would, however, be required to ensure the applicability of these indices for the region and the robustness of the derivation method. Water level requirements are also available for a number of distinct vegetation communities (Wheeler et al., 2004). These could be evaluated in conjunction with the existing study to define community succession under environmental change or different management scenarios.

f) Spatial examination of habitat availability

Although results from a distinct field survey are supportive of modelled values of riverine physical habitat availability, the study would benefit from further validation. The disaggregation approach of one-dimensional values to profiles across cross-sections is considered robust when projecting impacts for the reach as a whole. However, the method does not allow for local variations and spatial analysis, and the effect of weed growth on the vertical velocity profile is not accounted for. Although not applicable for assessment of the impacts of climate change, further field studies of habitat availability under different flow conditions would elucidate spatial differences and responses in habitat availability.

g) Testing other hydrological models

The detailed and high resolution data produced in this study provide a platform to test other integrated hydrological/hydraulic models. The hydrological modelling systems identified in this study's model review (Section 4.2.1) could provide a starting point for a more comprehensive review. With a complex geology and groundwater/surface-water interaction, the performance of different models may be assessed at the site scale. Such comparisons are useful to a wide audience due to model license fees, accessibility and availability.

h) Assessment of different management scenarios

The MIKE SHE / MIKE 11 and physical habitat modelling system employed in this study could be used to simulate the impacts of different management strategies. In developing adjustments to the existing management regime, weed cutting strategies with different severities, spatial application and timing may be modelled. The impacts of a range of weed cut options upon wetland water levels and available physical habitat could be

assessed under both current and scenario climates, plus other potential environmental changes such as abstraction. The addition of a nutrient or contaminant transport module would further aid impact assessment for particular species and communities. A range of selected management responses may be defined for different scenarios, creating a resource to aid conservation managers in adapting to system changes.

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